Role of the Estuary in the Recovery of Columbia River Basin Salmon and Steelhead: An Evaluation of Selected Factors on Population Viability

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PREFACE TO MAY 5 DRAFT

This May 5, 2004 version of the Estuary Technical Memorandum was prepared for the Collaboration Process on the Biological Opinion and includes the following major changes:

- Reorganization of the report
- Addition of a new section on other considerations in evaluating the role of the estuary.
- Addition of an Executive Summary
- An enhanced discussion of limiting factors from the perspective of each ESU.
- Reduction in duplication of material.
- Inclusion of a discussion of northern pikeminnow predation
- A revision of the section that rates factors. We developed a more formal rating system and simplified this section.

EXECUTIVE SUMMARY

The primary purpose of this report is to evaluate the potential of selected factors associated with the estuary to improve viability of listed anadromous salmonids in the Columbia River Basin. In addition, we examine the role of the estuary in the viability of anadromous salmonid populations, review what is known about salmon in the Columbia River estuary (which includes the plume), and examine how changes in selected factors associated with the estuary have potentially affected salmon. This evaluation was conducted in support of efforts by NOAA Fisheries to develop a Biological Opinion on the effects of the Federal Columbia River Power System (FCRPS) on listed anadromous populations in the basin.

For the purpose of this review, we defined the Columbia River estuary broadly to encompass the entire continuum where tidal forces and river flows interact, regardless of the extent of saltwater intrusion. The upstream extent of the estuary is Bonneville Dam and the downstream extent is defined to include the plume. Geomorphic features, ecological functions, and physical characteristics vary broadly within this area and give rise to a mix of habitats that the juvenile salmon can potentially occupy. Habitats vary throughout this region based upon site scale (e.g. depth, temperature, vegetation type, and substrate type) and landscape scale (e.g., connectivity, location within the estuary, shape, and size) attributes. Throughout the entire estuary, the distribution and quality of habitats has been affected (and continues to be affected) by a variety of anthropogenic (e.g., urbanization) and natural changes (e.g., climate change).

Our understanding of the role of the estuary in the life history and ecology of salmonid populations has changed considerably. Initial perspectives were driven by the perception that the major factors affecting salmon occurred in freshwater. Scientists became increasingly aware that non-freshwater factors had an important influence on numbers of returning adult salmon. This shifted attention to the role that the estuary and ocean played in salmon population fluctuations. The estuary came to be viewed as a bottleneck or limiting factor to the numbers of adults that could be produced. In more recent years, the estuary has been increasingly viewed as part of the salmon life cycle rather than as a place that salmon needed to avoid.

The approach adopted to define recovery needs of anadromous salmonids is based upon the concept that salmonids are comprised of populations or discrete breeding units that vary with respect to many attributes such as spatial and temporal use of habitats. NOAA Fisheries defines the status of anadromous salmonids based upon the viability of populations or groups of populations (ESUs) over long time frames. For populations to recover, the risk that they will go extinct needs to decline. Four performance criteria (VSP) are used to define viability: abundance, productivity, spatial structure, and diversity. Levels of these attributes in aggregate define extinction risk or persistence of the population. Estuaries help contribute to the viability of salmon populations by contributing to the range of places salmon can use (*spatial structure*), providing support for the life history strategies to use these places (*diversity*), and providing habitat capacity to produce successful recruits (*abundance* and *productivity*). Although all four VSP are critical to recovery, we suggest that the concepts of spatial structure and diversity are especially critical portion of the role of the estuary.

Ideally, we would like to be able to link factors in the estuary to their potential to affect the viability of each listed population. However, because we do not have specific, empirical information describing estuarine habitat use at the population level, we used an alternate approach. Specifically, we evaluated effects of candidate factors on an ESU based upon the life history type of that ESU and how different life history strategies associated with each ESU were affected by the factor. As each ESU is comprised of a bundle of populations, we can then infer responses of populations based upon what we predict will occur for the ESU.

Each listed ESU in the Columbia River basin (the Lower Columbia River coho salmon ESU was also included) was classified as either stream type or ocean type based upon characteristics of the juvenile outmigrants. When viewed over long time scales, most members of ocean type populations migrate to sea early in their first year of life after spending only a short period (or no time) rearing in freshwater. Most members of stream type fish migrate to sea after rearing for more extended periods in freshwater, usually at least a year. Thus, ocean type fish tend to spend longer periods in ocean habitats compared to stream type populations.

Individual members within each population employ a variety of alternative life history strategies or approaches to using available habitat. A life history strategy is a general approach to using available habitats, including the estuary. We used two attributes, size of fish at estuarine entry and time of estuarine entry, to define 6 general life history strategies that are common to all anadromous populations: 1) early fry, 2) late fry, 3) early fingerling, 4) late fingerling, 5) subyearling, and 6) yearling. Although each life history type produces some of each strategy, the relative proportion varies with life history type. Ocean type populations are dominated by the fry and fingerling strategies while stream type populations are dominated by yearling strategy.

Of the possible estuarine factors that could potentially viability, we considered the effects of four factors on salmon in the Columbia River estuary: flow, predation, habitat, and contaminants/toxics. These four were selected from a larger list of factors affecting salmon in the estuary based upon whether: 1) a significant change in the factor from historic conditions was evident, 2) the factor could potentially affect population viability, and 3) there was quantitative data available that could be used to analyze the affect of the factor (within the time we had been allotted). The factors that satisfied these criteria and were included in this analysis are water flow, availability of salmon habitats, toxics, and predation (Caspian terns). A brief evaluation as to how the factor has changed and how the factor could affect population viability was conducted based upon existing data and analyses. These analyses were based solely on impacts on juvenile life stages and did not include potential impacts on adults during their return migration. From these overviews, we developed a series of hypotheses or principles about each factor that helped guide how we rated their relative importance for each ESU.

It is important to note that factors not selected for our analyses may also have significant affects on salmon population viability; they were not included because they did not meat our selection criteria. For example, we expect that water temperatures have warmed from historic levels which could affect metabolic processes of both salmon and their predators. These changes could change mortality rates. Further, warmer water temperatures may exclude some habitat from use by juveniles during part of the year.

Flow is a fundamental factor affecting characteristics of salmon and their habitat in the estuary and plume. The interaction of flow and tides with the land creates and maintains estuarine habitat. Large scale changes in flow occur as a result of spatially explicit interactions of short and long term climate cycles (ENSO and PDO, respectively) with the watershed. Operations of the Federal Hydropower system (e.g., generation of electricity, flood control, and irrigation) have had significant affects on attributes of flow. These include a reduction in the mean annual flow, reductions in the size of the spring freshets, an almost complete loss of overbank flows, and changes in timing of ecologically important flow events. The hydrological changes, along with floodplain diking, represent a fundamental shift in the physical state of the Columbia River ecosystem. Major changes in the estuary resulting from flow alterations that are especially relevant to salmon include a loss of vegetated, shallow water habitat and changes in the size, seasonality, and behavior of the plume. Such changes potentially have significant consequences for both expression of salmonid diversity and productivity of the populations. In particular, because the changes in habitat are most pronounced in shallow water areas, effects on the ESUs and life history strategies (the fry and fingerling strategies) that use and depend upon these shallow water areas is most significant.

The location and types of habitats present in the Columbia River Estuary have been substantially altered from historic conditions. Although the entire estuary has not yet been surveyed, the main changes that have been identifed in the estuary have been a major loss of emergent marsh, tidal swamp, and forested wetlands and shifts in organic matter important to estuarine food webs; changes in the plume have also occurred.

Shallow water dependent life history strategies (fry and fingerlings) have been most affected by the loss of the vegetated habitat types while larger life history strategies have been most affected by changes in the plume. Alterations in attributes of flow and the construction of dikes and levees have caused these changes. Diking is a significant change primarily because it severs the connection of the habitat with the river so it provides no direct (use) or indirect (export of organic matter for food webs) benefits to the fish.

Exposure to water borne and sediment associated chemical contaminants has the potential to affect survival and productivity of both ocean and stream-type stocks in the estuary. Stream-type ESUs are likely to be most affected by short-term exposure to waterborne contaminants such as current use pesticides and dissolved metals. These chemicals can disrupt olfactory function and interfere with such behaviors as capturing prey, avoiding predators, imprinting and homing. Ocean-type ESUs may also be exposed to these types of contaminants, but will also be affected by persistent, bioaccumulative toxicants such as PCBs and DDTs, which they may absorb during their more extended estuarine residence. Consequently, we expect that the impact on ESUs exhibiting the ocean life history type will be more significant.

Predation is a major source of mortality of all salmonid populations. Although many predator prey interactions in the Columbia River estuary appear to have changed from historic conditions, we have little quantitative data on how this has affected population viability for most predators. One exception to this is Caspian tern predation which has significantly increased due to a recent change in nesting habits of the birds. The main impact of tern predation is on ESUs with stream type life history types, especially steelhead. This is primarily because the dominant migratory periods employed by salmonids with a stream type life history most overlap with the nesting period of the terns. Improvements to productivity of populations by managing terns would be expected to benefit stream types ESUs especially, although lesser benefits to other salmonid ESUs in the basin should also occur.

To rate the importance of each the four factors, we developed a rating system that ranked each factor as having a high, medium, or low ability to improve the status of anadromous salmon populations. We drew inferences about how a factor affects an ESU based upon the life history type of that ESU and how we believed the factor would affect the life history strategies that characterized that life history type. Thus, the limiting factors for all stream type ESUs were ranked similarly while those for ocean type ESUs were ranked similarly. Ratings were developed by considering each factor relative to other estuarine factors within an ESU; ratings were not considered with the context of non-estuarine factors such as tributary habitat.

The rating system consisted of two levels and each level asked two questions. The level 1 screens evaluated ***if*** the factor was likely a concern for an ESU based upon its affects on VSP and change in the factor from historic conditions. The level 2 screens asked ***how*** the factor affected an ESU based upon where the affects occurred. Each questions was evaluated for each factor for each ESU based upon whether the ESU was ocean or stream types. Scoring was done using guidance from the principles/hypotheses developed in the discussion of the limiting factors. Because of limitations in our knowledge base, we aggregated the estuary into two zones: river mouth to Bonneville and the plume. Within the portion from river to Bonneville, we also only differentiated two habitat types: shallow and deep. Each cell in the matrix was either scored as a yes (+1) or no (0) with two exceptions: 1) abundance and productivity which were given a 2 score, and 2) toxics in deep and shallow water which each could be scored a 2 if there was effects from both water borne and sediment associated toxics. Because this changed the maximum possible score for each factor, the final rating was computed as the ratio between the assigned score and maximum possible score.

For stream type ESUs (e.g., upper Columbia River chinook salmon and Snake River steelhead), the primary factors affecting population viability were tern predation and flow. The increase in tern predation from historic conditions is not a direct result of Hydropower Operations while flow changes are mostly a direct effect of dam operations. Tern predation was ranked in the medium category. We concluded that abundance of the main life history strategies associated with stream type ESUs were significantly affected; the main effects of this factor are on abundance and productivity. Flow changes were ranked medium. The main affects of flow on this life history type is on the dominant life history strategies (e.g., yearlings) in the plume. Toxics and habitat were ranked low for stream type ESUs because the main life history strategies associated with this ESU do not occupy the habitats where the these factors have the most significant impacts.

For ocean type ESUs (e.g., Lower Columbia River chum salmon and Snake River fall chinook), flow and habitat were rated as having a high ability to affect population viability. As noted above, flow changes in the basin are primarily a result of dam operations whereas habitat changes are a function of both hydropower operations and other, non-hydro issues, notably the construction of dikes and levees. The combined affect of flow and habitat changes on estuarine habitat has been to reduce the amount of shallow water habitat (especially vegetated) and disrupt organic matter inputs from these vegetated habitats. The dominant life history strategy of ocean type chinook salmon extensively use shallow water habitat which is where the main flow and habitat changes have occurred. Tern predation was considered to have a low affect on this life history type because terns do not prey significantly on fry and fingerling strategies (the dominant ones associated with this ESU). Contaminants received a medium score. Both water borne and sediment contaminants can affect fry and fingerling life history strategies in shallow water areas.

From the perspective of the estuary, we conclude that population viability of stream type ESUs is most affected by tern predation and flow while flow and habitat most affect ocean type ESUs. At this time, we do not know how much of a change in each factor is required to affect relevant ESUs. Based upon available information, we hypothesize that the greatest opportunity to affect ESUs in the Columbia River basin by the manipulation of estuarine factors is with restoration of lost, shallow water habitat. These actions will primarily affect ocean type ESUs and the shallow water dependent strategies of stream type ESUs. This is because there is a strong linkage between the fry and fingerling life history strategies, which dominant ocean type ESUs, and shallow water habitat. Thus, the main affect on ocean type ESUs is on abundance and productivity while spatial structure and diversity of stream type ESUs is most affected. There is a large amount of vegetated shallow water habitat that has been lost due to diking. Restoration of the shallow water habitat can also be done without changing hydro operations, which can have other, unintended consequences, such as an increase in gas bubble disease. However, the design of restoration strategies for shallow water habitat must consider flow regimes because of the strong effects of flow on functions of this habitat type. We expect that studies now underway will provide greater insight into how much change in shallow water habitat is both possible and needed to affect population viability.

CONTENTS

INTRODUCTION

Since 1991, 12 different populations of anadromous salmonids that reproduce in the Columbia River Basin have been listed as threatened or endangered under the Endangered Species Act (ESA) of the United States. These populations include steelhead, chum, chinook, and sockeye populations that spawn from the upper Snake River Basin to tributaries of the lower river below Bonneville Dam. Every subbasin of the Columbia that is currently accessible to anadromous salmonids contains at least one threatened or endangered population. The Federal Columbia River Power System (FCRPS) has had a variety of well documented impacts on anadromous salmonids in the basin including the loss and degradation of spawning and rearing habitat and increased mortality of upstream and downstream migrating fish during passage at hydroelectric facilities (ISG 2000). As a result, efforts to recover these populations at risk have almost exclusively focused on identifying and modifying risk factors directly associated with the large hydroelectric dams constructed throughout the basin.

Increasing attention is now being focused on other effects of the hydropower system and non-hydropower related issues in the decline and recovery of salmonids in the basin. One area that encompasses both of these factors is the condition and availability of habitats in the estuary. The growing recognition that the estuary has a role in the recovery of Columbia Basin salmonids represents a significant departure from previous management efforts in the system. There are several developments which appear to be responsible for this shift. First, legislation passed in 1996 by Congress that amended the Power Act requiring the Northwest Power Planning Council to consider the effect of ocean conditions on fish and wildlife populations when recommending hydropower mitigation projects for the Columbia River Basin. This legislation focused new attention on the estuary, plume and coastal ocean habitats.

Second, life stage risk and sensitivity modeling analyses of Columbia River salmon populations by Kareiva et al. (2000) and McClure et al. (2003) suggest that additional actions beyond passage improvements are needed to recover salmonid populations. Two life stages identified as sensitive to perturbations included the first years spent rearing in the river, estuary and ocean. Kareiva et al. (2000) concluded that the maximum potential to contribute to anadromous salmonid recovery was associated with these life stages.

Third, scientific perspectives of the life history and ecology of anadromous salmonids have shifted in recent years. Previously, habitats and life stages important to salmon were considered in isolation with the goal of identifying single limiting factors restricting salmon production. We now recognize that marine, estuarine and riverine environments are each components of an extended salmon ecosystem that cannot be treated independently (ISG 2000; Bisbal and McConnaha 1998). Thus, the estuary and Columbia River plume are part of the continuum of landscapes all juvenile and adult anadromous salmonids use that originate from the Columbia River Basin. They are the connection between freshwater and marine habitats and are used by all life stages to some degree for feeding, refugia from predators, and physiological transition (McCabe et al. 1983, 1986; Bottom and Jones 1990). Moreover, recent research shows that decadal scale regime shifts in climatic and oceanic conditions can produce long term changes in salmon production across the entire North Pacific Ocean (Francis and Sibley 1991; Beamish and Bouillon 1993; Mantua et al. 1997). Such shifts in the production regime along with natural variability must be taken into account when developing appropriate recovery goals, actions and expectations for Columbia River salmonids.

Finally, our understanding of the functions of habitats in the persistence of salmon populations has evolved in recent years. We now recognize that habitats cannot be valued simply on the basis of their role in producing fish biomass or abundance (Bottom 1997). Instead, diverse, high quality habitats and the expression of life history strategies based upon use of these habitats are directly linked to salmon population viability (i.e.., persistence) over long time scales (McElhany et al. 2000). These linkages were explicitly recognized by the ISG for the NWPPC who concluded that estuary/ocean dynamics helps to control salmon productivity and that salmon biodiversity (including the diversity of estuarine life histories) minimizes the effects of fluctuations in ocean and presumably freshwater conditions (ISG 2000). Alterations and reductions of estuarine habitats implies the changes (such as elimination) in estuarine dependent strategies, which has implications for population viability.

The challenges of identifying, designing, implementing and evaluating recovery actions in the estuary are significant, in part because we know little about the estuary and the salmon that use the estuary. While ongoing research efforts will significantly upgrade our knowledge base in upcoming years, much of what we now know is conceptual or based on research from other areas such as Puget Sound. Estuary restoration at any scale is a challenge and in the Columbia River estuary it is an especially daunting challenge because of the massive size of this system. Further, the estuary is among the most heavily modified portions of the basin (Thomas 1983) due to the long history of coastal development and the cumulative effects of flow regulation, habitat modification in the estuary and other changes upriver which have altered sediment transport and salinity regimes in the system (Simenstad et al. 1992; Weitkamp 1994). In the last 100 years, these and other changes have decreased the amount of some types of wetland habitats in this region by as much as 70% from historical levels (LCREP 1999).

The primary purpose of this report is to evaluate and rank candidate limiting factors in the estuary and plume with respect to their potential to affect population status or suppress population specific recovery. This evaluation was conducted in support of efforts to develop and implement recovery actions in the basin. Our major objectives were to: 1) examine the role of the estuary and plume in the viability of anadromous salmonid populations, 2) review what is known about salmon and their use of estuarine and plume habitats, 3) examine how several factors affect salmon and their habitats in the estuary and plume, and 4) evaluate candidate factors associated with the estuary and plume with respect to their potential to improve the condition of listed populations in the Basin (Lower Columbia River coho salmon was included as well).

THE COLUMBIA RIVER ESTUARY

An estuary is generally defined as a semi-enclosed coastal body of water with a free connection to the open ocean in which salt water is diluted with runoff from the land (Pritchard 1967). For the purpose of this review, however, we defined the Columbia River estuary more broadly to encompass the entire habitat continuum where tidal forces and river flows interact, regardless of the extent of saltwater intrusion. During low river flows, salt-water enters deeper channels only as far as River Mile 30 (RM-30) although tidal influence extends upstream all the way to Bonneville Dam (RM-145) (Figure 1). The upstream bound of the estuary is Bonneville Dam while the "downstream" boundary includes the plume.

The estuary can be further divided into different zones based upon various attributes such as geomorphic features, ecological functions, and physical characteristics and each zone can be further subdivided into different habitat types and features (Figure 1). Here, we define four zones. First, from approximately RM-45 to RM-145 is a long tidal-freshwater zone (referred to as the tidal river zone) where the river is constrained to a simple deep channel and there is only narrow fringe of intertidal habitat. Second, between Tongue Point (RM-18) and upper Puget Island (RM-45) there is a large estuarine mixing zone (referred to as the estuarine mixing zone) where mean salinities range from 0-15 parts per thousand (in deep channels only). At this point, the estuary widens into a series of complex (based upon vegetation and shoreline geomorphology) islands, forested and emergent wetlands, and low-salinity bays (Grays Bay and Cathlamet Bay). Third, from Tongue Point to the river mouth is a high-energy zone from the river mouth to Tongue Point where the salinity gradient increases to more than 30 parts per thousand at the river entrance (referred to as the lower estuary). A major feature of this zone is the pair of shallow, peripheral bays (Baker Bay and Youngs Bay) with expansive intertidal flats that occur along either side of the lower estuary.

The fourth zone is the Columbia River plume, the final transitional zone the salmon must occupy before they are fully entrained in oceanic habitats (Figure 1). The Columbia River exerts substantial influence on the physical properties of the northeast Pacific Ocean, accounting for 60 to 90% of the total freshwater inflow into the ocean between San Francisco Bay and the Strait of Juan de Fuca (Fox et al. 1984). During high flows, the plume front is readily visible as a sharp interface between sediment-laden river water and the clear ocean. The river plume is generally defined by a reduced-salinity contour near the ocean surface of 31 parts per thousand. Its geographic position varies

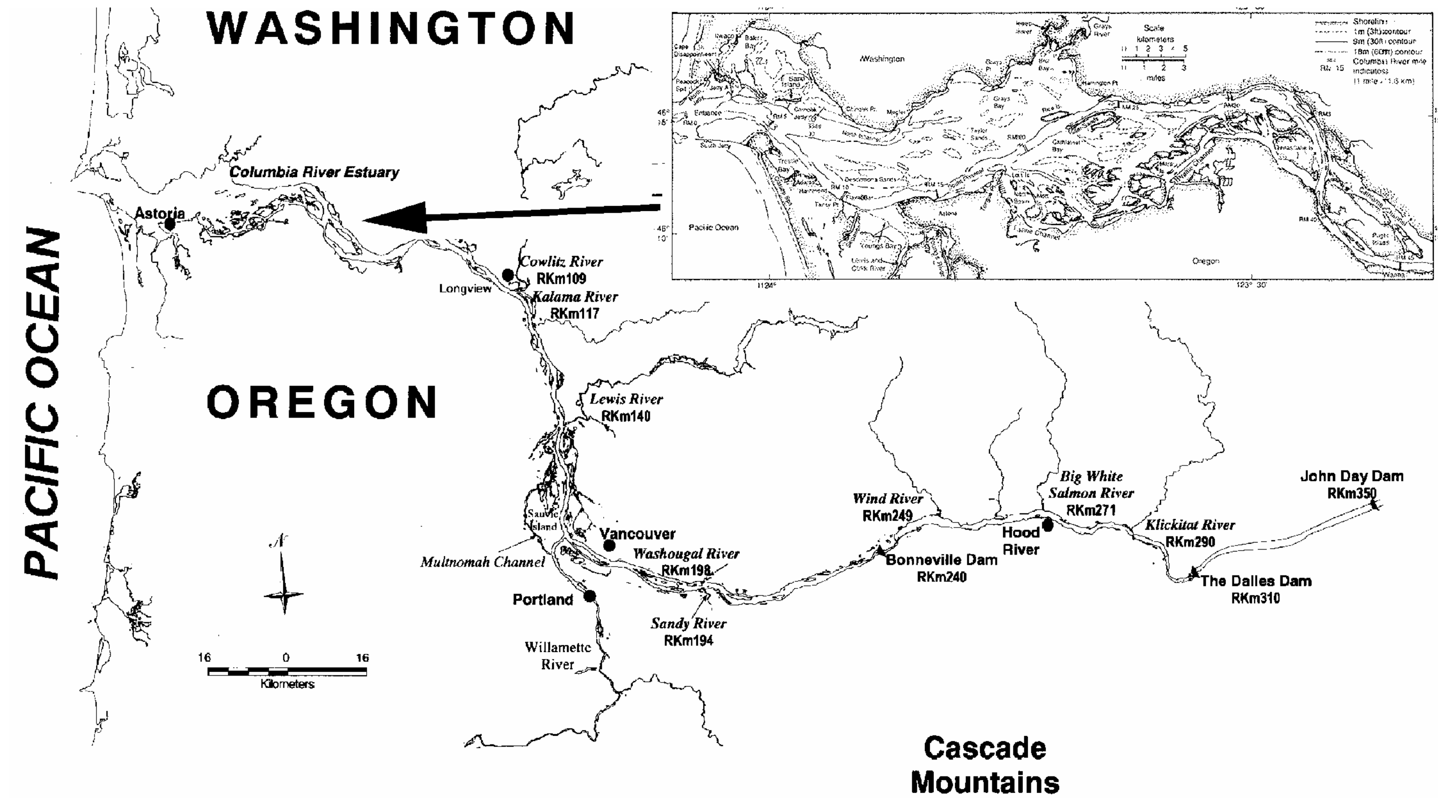


Figure 1. The Columbia River estuary extends from the upper extent of tidal influence at Bonneville Dam (RKm 240) through the oligohaline zone of the river mouth into the coastal zone of the plume in the Pacific Ocean. Inset shows the estuarine study region (to RKm 75) for the Columbia River Estuary Data Development Program (Simenstad et al. 1990a).

greatly with seasonal changes in river discharge, prevailing nearshore winds and ocean currents. During summer months, the plume extends far to the south and offshore along the Oregon coast; during the winter it shifts northward and inshore along the Washington coast. Strong density gradients between ocean and plume waters create relatively stable habitat features where organic matter and organisms are concentrated. Recent evidence suggests that this offshore extension of the estuary serves as another important habitat for outmigrating juvenile salmon.

Within each of these zones, including the plume, is a mix of habitats that the juvenile salmon can potentially occupy. Habitats can be classified based upon site scale (e.g. depth, temperature, salinity gradient, vegetation type, and substrate type) and landscape scale (e.g., connectivity, shape, and size) attributes. The functions of these habitats for juvenile salmon and steelhead depend upon how these attributes, in aggregate, affect the accessibility of the habitat to the fish and its quality (Simenstad and Cordell 2000). Table 1 presents a general classification of major habitat types within the estuary below RM 46, after Thomas (1983) and Johnson et al. (2003); classification systems have not been developed for other parts of the estuary.

Table 1. Major types of estuarine habitats and some of their important attributes in the Columbia River below RM 46 (partially after Thomas 1983 and Johnson et al. 2003).

Major habitat types	Important Attributes
Tidal swamps	Vegetation is mostly shrub and woody species. Higher elevations. Low water velocities
Tidal marshes	Dominant vegetation varies. Includes emergent marshes. Tidal channels often present. Depths generally range from MLLW to above MHHW. Low water velocities.
Tidal flats	Ranges in depth between MLLW and 6 ft below MLLW. Usually not vegetated.
Medium deep	Depth range: 3-18 ft. Mostly associated with medium sized and larger channels. Higher water velocities.
Deep	> 18 feet in depth. Mostly associated with main channel. Higher water velocities.

ROLE OF THE ESTUARY IN VIABILITY OF SALMON POPULATIONS

Although we have been studying Pacific salmon for 100+ years, we have been investigating the role of the estuary in salmon life history and ecology for less than half that time. Early researchers focused on the freshwater phase of life, primarily because it was assumed that numbers of returning adult salmon were a function of conditions occurring in freshwater habitats (e.g., Neave 1953; Walters et al. 1978). It was not until the 1950's and 1960's that researchers began to recognize the possibility that non-freshwater factors also had a role in determining numbers of returning adults (e.g., Manzer and Shepard 1962; Gilhousen 1962). The development and analysis of long-term data sets on salmon production suggested freshwater conditions could not by themselves adequately explain variability in numbers of returning adults (e.g., Salo and Bayliff 1958; Hunter 1959; Gilhousen 1962; Parker 1968; Peterman 1978). This stimulated research beginning in the late 1960's in Puget Sound, British Columbia, and Alaska, on the estuarine and early marine periods of life of juvenile salmon (e.g., Stober and Salo 1973; Kaczynski et al. 1973; Reimers 1973; Mason 1974; Bailey et al. 1975; Salo et al. 1980; Healey 1979, 1980).

Several studies were particularly instrumental in directing attention at the estuarine and early marine periods of life. Parker's analyses of pink salmon population dynamics in a Central British Columbia river system suggested that there was a period of very intense mortality that occurred during early marine life (Parker 1968). He concluded in an ensuing study that predation was the primary factor causing this mortality (Parker 1971). In Puget Sound, the work by Kaczynski et al. (1973) on the feeding ecology of small pink and chum fry in littoral areas was critical because it demonstrated that these small fish were closely linked to shallow water habitats during their early marine life in a way that had not been previously appreciated. Studies in the Nanaimo River, B.C., confirmed this close linkage between shoreline areas and chum and pink fry with the discovery that the major prey items of these fish originated from organic matter produced within this zone (Healey 1979; Sibert et al. 1979). This represented a significant departure from traditional views of marine food webs where organic matter was assumed to come from pelagic sources.

During this early period of estuarine research, the Columbia River estuary was largely ignored by investigators. Although estuarine related research has occurred throughout the Pacific Northwest since the late 1960's, use of the Columbia River estuary (and plume) by juvenile salmon was not studied until recently. It is remarkable that despite the importance of Columbia River salmon in the Pacific Northwest and the large size and diversity of this estuarine system, empirical knowledge about how salmon use this estuary is lacking compared to other estuarine systems in the Pacific Northwest.

Production vs. Population Perspectives of Salmon in Estuaries: A Shift in Paradigms

One reason why estuarine related research on salmon evolved so late is that for decades salmon species were primarily assumed to be regulated by density dependent factors in freshwater (Bottom 1997; Bottom et al. 2001); more adults were expected to return simply as a result of an increase in the number of eggs and fry. The estuarine and ocean environment were considered unimportant or irrelevant. Thus, a major goal of early salmon research was to understand freshwater sources of mortality so that they could be more easily manipulated and so more adults produced (Bottom 1997). Salmon were generally viewed at this time as simply another agricultural product to be managed for the benefit of people (Bottom 1997). Such a production perspective viewed salmon largely as a crop and the output of this crop was defined as short term changes in numbers of fish harvested or reproducing. Initial theoretical models of salmon population dynamics such as spawner-recruit relationships (e.g., Simon and Larkin 1972) and the concept of maximum sustained yield evolved from the perspective that salmon species were controlled by density dependent factors.

The production view of salmon and their environment has been intertwined with our use of hatchery production. Hatcheries have been used for much of the last 100 years to increase numbers of returning adults to compensate for the fact there has never been enough salmon to go around due to habitat destruction, high demand for salmon, harvest, and expanding human population (Lichatowitch and McIntyre 1987). Hatcheries evolved from a density dependent philosophy that more adults would result in direct proportion to the additional number of eggs that survived (Lichatowitch 1999). They were a direct outgrowth of the production view of salmon because they focused on bypassing portions of freshwater life where the most significant sources of mortality were believed to occur. The goal of hatchery technology has always been to control more and more of the fish's freshwater life in order to bypass as much mortality as possible.

The continued failure of hatchery production to maintain or increase salmon returns raised new questions about whether passage through estuaries and the ocean might be critical to determining numbers of returning adults. Estuarine and early marine life rapidly came to be regarded as ***the*** "critical period" of high mortality that significantly affected overall survival rates and adult returns (Kaczynski et al. 1973; Ricker 1976; Peterman 1978; Healey 1980; Nickelson 1986; Pearcy 1992) and as a bottleneck to salmon productivity where mortality was high (Parker 1968; Bax 1983).

In the 1970's, the prospects that expansions in hatchery production would be used to meet continued high demand for salmon generated concern that there was a limit or carrying capacity to estuarine environments. Simenstad et al. (1978) expressed concern that the carrying capacity of the Hood Canal estuarine environment for pink and chum salmon would be exceeded if enough hatchery fish were released. Bailey et al. (1975) suggested that releases of hatchery pink and chum in Alaska coastal areas would result in fewer adult returns than expected by exceeding carrying capacities of these habitats. Studies were initiated to estimate the quantities or carrying capacity of hatchery fish that could be supported by estuaries (e.g., Reimers et al. 1979) and to find the optimum conditions in the estuary that would maximize production. Because some managers questioned if salmon were really estuarine dependent, further studies were conducted to ascertain if the estuary could be bypassed altogether and so render the whole issue of carrying capacity in this environment moot (Macdonald et al. 1988; Solazzi et al. 1991).

In contrast to the production perspective of salmon is the *population* perspective (Bottom et al. 2001). The population view was to some degree an outgrowth of studies by Willis Rich on Columbia River salmon (e.g., Rich 1939), evaluating scale patterns of juveniles passing through the Columbia River estuary. He found a variety of patterns of estuarine use with some fish present nearly year round in the estuary. Fish also exhibited a wide variety in the time at which they arrived in the estuary, the amount of time they were spending there, and the size at which they arrived (Rich 1939).

To explain his results, Rich suggested that salmon species were comprised of a set of geographically discrete, self-perpetuating populations. Within each population, a range of behaviors was exhibited that were defined by the particular set of conditions found in the full range of spawning and nursery areas used. He concluded that there was not a unique or singular way for a salmon species or population to use the estuary or any other habitat but instead populations employed a diversity of approaches. Groups of salmon became locally adapted to the conditions they experienced. Rich proposed that use of the estuary or any other habitat the fish were occupying (e.g., when the fish arrived, size at arrival, how long they resided) depended upon which population the fish had come from. His view of the Columbia River estuary was that it was a mixing ground of fish from many different sources. The view that salmon are comprised of populations has become a fundamental tenant of salmon management, biology and research as it evolved into the stock concept whereby salmon are managed as stocks or subunits of the species (e.g., Simon and Larkin 1972).

A Conceptual Framework for the Role of the Estuary in the Recovery of Anadromous Salmonids

In order to rebuild depleted anadromous salmonids throughout the Pacific Northwest, an approach to defining recovery needs of salmonids is needed. NOAA Fisheries defines the status of anadromous salmonids based upon the ability of populations or groups of populations (ESUs) to persist or remain viable over long time frames. In general, this means that for populations to recover, the risk that they will go extinct needs to decline (McElhaney et al. 2000). This approach to recovery follows directly from the principle that salmon species are comprised of populations. NOAA Fisheries uses four performance criteria (VSP) to define viability (McElhaney et al. 2000): abundance, productivity, spatial structure, and diversity. Levels of these attributes in aggregate define extinction risk or persistence of the population.

Abundance is simply defined as some measure of the number of members in the population such as numbers of spawners or returning adults while productivity is the rate of growth of the population over some time interval. Both are straightforward concepts that suggest that populations that have lots of members and have a positive population growth rate are more likely to persist then populations that do not have these characteristics. Evidence clearly suggests that estuarine habitats provide capacity that contributes to the abundance and productivity of salmon populations (e.g., MacDonald et al. 1988). For example, Reimers (1973) demonstrated that for the one brood year of chinook salmon that he studied in the Sixes River, most adult returns originated from fish that made the most extensive use of the estuary. Magnusson and Hilborn (2003) similarly concluded that survival to adult return of hatchery chinook salmon populations in coastal environments was directly and positively correlated with the condition of the estuary.

Spatial structure refers to the geographic distribution of individuals in the population and the processes that generate that distribution and diversity refers to the variability in traits exhibited by salmon both between and within populations. Both spatial structure and diversity reflect the strategies or approaches that salmon populations have evolved to cope with the extensive environmental variability they experience throughout their lives (Healey 1991; Healey and Prince 1995). Although conservation of life history diversity and spatial structure of habitats is an emerging paradigm in recovery and management of Pacific Salmon (e.g., McElhaney et al. 2000; Issak et al. 2003) and other fish species (e.g., Gresswell et al. 1994), these two VSP criteria are the least well understood and their application in salmon recovery is a considerable challenge. For example, within Puget Sound, abundance and productivity goals have been developed for many populations but no such goals exist for spatial structure and diversity. One reason is that while abundance and productivity can be measured over many time scales, spatial structure and diversity are most relevant at long time scales. Although all four VSP parameters are critical to the viability of salmon populations, the role of spatial structure and diversity in salmon recovery are least understood and are important to an understanding the role of the estuary in salmon recovery; we consider these concepts within the context of the estuary in greater detail below.

The spatial structure of a salmon population refers to the distribution of members in space. Because the spatial structure of a population fundamentally depends upon the distribution, diversity, configuration, and quality of habitats used by the salmon, these types of habitat metrics can be used to measure or assess spatial distribution. Levin (1960) proposed principles of metapopulation dynamics to describe how groups of populations (i.e., species) interacted with habitat; these principles have attracted considerable interest in salmonid conservation (Rieman and Dunham 2000; Issak et al. 2003). These same metapopulation principles can also be applied at a population scale and used to describe the relationships between the spatial geometry of habitats and the dynamics and long-term persistence of a population.

According to metapopulation dynamics, persistence of a population in a variable environment will depend in part upon the spatial geometry of suitable habitat, including numbers, qualities, and quantities of habitat patches occupied, patterns in the use of patches, when patches are occupied, and the ability of population members to colonize and use habitat patches. At any one time, there are a wide variety of habitats occupied by members of a population, although not all suitable habitats that are available will be occupied.

Diversity consists of the variability in traits exhibited by salmon. Such variability is expressed both within and between populations in a wide variety of traits including body size, fecundity, timing of life history events, location of spawning, residence time in various habitats, habitat use, size at age, age at maturity, ocean distribution patterns and physiological characteristics (Healey and Heard 1984; Tallman and Healey 1991; Taylor 1991; NRC 1996; Beckman et al. 2003; Miller and Sadro 2003; Ramstad et al. 2003).

Between populations, diversity reflects the local adaptations that different salmon populations have evolved to cope with the specific conditions that they experience (Beachum and Murray 1987; Taylor 1990; Roni and Quinn 1995; Quinn and Unwin 1993). Because of differences in spawning, rearing, and migratory environments, populations become genetically and phenotypically distinct (Burger et al. 1985; Tallman and Healey 1991; Quinn et al. 2000; Hodgson and Quinn 2002; Ramstad et al. 2003). *Within* populations, diversity represents a strategy to spread risks spatially and temporally in the face of large amounts of environmental variability.

Healey (1991) termed this type of diversity tactical in nature. While the factors that produce diversity within a population are unclear, it is likely an interaction of the genotype of the animals, where the fish emerged from the gravel, incubation environment, environmental conditions during spawning and early freshwater residence, extreme environmental events, micro habitat, and biological interactions (e.g., Taylor 1990; Healey and Prince 1995).

One way to conceptualize diversity is as a set of alternate life history pathways, strategies, or trajectories (Wissmar and Simenstad 1998) that individual members of a salmon population can follow. A strategy simply represents an approach to using the spawning, rearing, and migration habitats that are available to the fish in space and time. This variability in how salmon use habitats helps populations persist in the face of spatially and temporally fluctuations in the environment.

These trajectories can be bundled or aggregated into a more limited number of general trajectories based upon spatial and temporal patterns in use of habitats, although there is not a single accepted approach for defining strategies (e.g., Reimers 1973; Carl and Healey 1984; E. Beamer, SSC, personal communication). The number and success (i.e., survival) of fish using each of these trajectories varies and depends upon biological and environmental processes operating over multiple scales. During any particular time period (e.g., a particular ocean regime), there are trajectories that work best for a particular population while other trajectories will produce large number of recruits when conditions change.

The number and quality of trajectories or strategies provides a measure of the diversity of a population and its ability to persist over time. Differences in habitat use between population members can then be used as a way to distinguish trajectories. In addition, other phenotypic (and potentially genetic) differences (e.g., age at return and size at age) may occur between trajectories depending upon the nature of the specific conditions in the spawning, rearing, and migratory environments each trajectory experiences. A major factor affecting the number and quality of strategies present within a population will be the spatial structure of habitats that can potentially be used by the salmon (NRC 1996). If the habitats do not exist because of either natural or anthropogenic factors, then population members cannot use them and potentially these life history types can be eliminated from the population. In order for a population to use diverse habitats requires that the right life history types exist and the existence of the right history types depends on existence of the appropriate habitats.

The complex geographic distribution of populations and the alternate approaches to completing life cycles are not unique to salmon or anadromous species (e.g., Roughgarden et al. 1988; Sinclair 1988; Secor 1992; Able et al. 2003). Within the Alagnak River, Alaska, Meka et al. (2003) found that rainbow trout exhibited three life history patterns based upon their migratory movements within this river system. A similar diversity in movements within Yellowstone cutthroat trout was described by Gresswell et al. (1994). Although little genetic differentiation existed, they concluded that the range in variation in life history strategies within the cutthroat trout was an adequate basis for providing protection to each life history type.

Curry et al. (2002) found a range of tactics related to the use of freshwater and estuarine habitats within one riverine brook trout population. Recent research has revealed that striped bass exhibit a variety of life history approaches that can vary within and between populations in use of freshwater and estuarine landscapes (Secor 1992; Secor and Piccoli 1996). Many populations of marine fish also exhibit complex approaches to how they distribute themselves in space and time that are similar to the tactics exhibited by anadromous species (e.g., Roughgarden et al. 1988; Able et al 2003).

In summary, recovery of salmon populations requires that viability of the population increases over long time scales. Estuaries help contribute to the viability of salmon populations by contributing to the range of places salmon can use (*spatial structure*), providing support for the life history strategies to use these places (*diversity*), and providing habitat capacity to produce successful recruits (*abundance* and *productivity*).

USE OF ESTUARINE HABITATS BY COLUMBIA RIVER SALMONIDS

As mentioned previously, there have been few studies of habitat use of the Columbia River estuary by wild juvenile salmon and steelhead. Much of the estuary, especially the portion from Bonneville Dam to Puget Island, has not been studied at all. Since 2001, work has been conducted by NOAA Fisheries in the middle and lower estuary that will significantly upgrade our knowledge about how juvenile salmon and specific ESUs use the estuary. One factor that needs to be considered in any analysis of estuarine use by wild salmon and steelhead is the occurrence of hatchery fish. Because our ability to separate wild and hatchery fish captured in the estuary has been limited, and remains so even at present, many of the spatial and temporal patterns observed in historical data sets may apply to hatchery fish rather than wild fish (e.g., Dawley et al. 1985, 1986).

The only study that provides enough information to distinguish use of different estuarine habitats was by CREDDP (Columbia River Estuary Data Development Program). There are several limitations of this research. First, the work only occurred in the lower estuary. Second, because CREDDP infrequently sampled shallow water habitats off the main channel, potential use of these areas is largely unknown. In general, McCabe et al. (1986) found that subyearling chinook in shallow intertidal habitats of the Columbia River were smaller than subyearlings captured in deeper pelagic areas. Larger, yearling migrants spend little time in shallow estuarine habitats and more time in deeper channel sties (Bottom et al. 1984; McCabe et al. 1986).

Most of what is known about juvenile salmon use of the estuary concerns timing of fish passage through the estuary and was derived from seining studies conducted to recapture coded wire tagged (CWT) fish below Bonneville Dam. In the late 1970’s and early 1980’s, subyearling chinook salmon (these are all non-yearling fish combined) were present year round (Bottom et al. 1984; Dawley et al. 1986; McCabe et al. 1986). While in many of the years studied, overall peak abundance occurred from May to September, there were years when a bimodal distribution was observed (Dawley et al. 1985, 1986). There was also evidence that there were specific patterns in seasonal timing that particular populations exhibited that deviated from the general population- e.g., Lewis River (Dawley et al. 1985). Peak catches at Jones Beach, where much of the “estuary” timing work has been conducted, often were highly correlated with the timing of hatchery releases.

Although knowledge of estuarine habitat use in the Columbia River basin is limited, the information that does exist in combination with studies in other estuaries of the Northwest can be used to provide insight into how juvenile salmonids may use habitats in this large estuary. Estuarine research has demonstrated that juvenile salmon are generally distributed along a habitat continuum based upon water depth (Healey 1980; Levy and Northcote 1982; Simenstad et al. 1982; Bottom et al. 1984; Levings et al. 1986; McCabe et al. 1986; Miller and Sadro 2003). As fish size increases, depth of the water used by the fish increases; fish size can change as a result of growth that occurs in the estuary, growth in freshwater, or some combination of rearing in the two environments.

Based upon this size based model, the smallest juvenile salmon in the estuary (fry and fingerlings) will be primarily associated with the shallowest, peripheral, wetland type of habitat while the larger subyearlings and yearlings will be found in deeper pelagic areas. Coincident with the fish size/depth relationship, there is a general pattern for smaller salmon to spend longer in the estuary and for the larger, yearling migrants to spend less time.

APPROACH TO ANALYSIS OF FACTOR EFFECTS ON POPULATION VIABILITY

The overall purpose in this report is to evaluate selected factors in the estuary with respect to their potential to improve viability of listed populations of anadromous salmonids in the basin. We considered viability from the perspective of the four performance criteria used by NOAA Fisheries to evaluate population viability abundance, population growth rate, spatial structure, and diversity (McElhany et al. 2000). Ideally, we would like to link factors in the estuary to their potential to affect the viability of each listed population. However, because we do not have specific, empirical information describing estuarine and plume habitat use by anadromous populations in the Columbia River estuary and plume, we used an alternate approach whereby effects of candidate factors were linked to viability of an ESU based upon the life history type of each ESU and how a factor affected the distribution and quality of life history strategies associated with each life history type. As each ESU is comprised of a bundle of populations, we can then infer responses of populations based upon what we predict will occur for the ESU.

Defining Life History Type and Life History Strategy

We defined each ESU according to whether it was either stream type or ocean type. The division of salmon populations into these two **major life history types** was originally suggested by Gilbert (1912) to distinguish individual fish that emigrated to the ocean as subyearlings (ocean type) and those that remained in freshwater for at least a year prior to emigration (stream type). The concept has mostly been applied to chinook salmon populations (Myers et al. 1998), as recent genetic analyses of chinook populations have found that that stream and ocean type populations are distinct evolutionary lineages that can be distinguished genetically (Carl and Healey 1984; Healey 1991; Teel et al 2000; Rasmussen et al. 2003). In recent years, the terms stream and ocean type have been increasingly applied to other species as well.

Although stream and ocean type fish can be distinguished from each other by differences in ocean distribution patterns (Healey 1991), the major distinguishing feature used to separate these two life history types is characteristics of the juvenile outmigrants (Table 2). Ocean type populations are generally (but not exclusively) composed of members that migrate to sea early in their first year of life after spending only a short period (or no time) rearing in freshwater. Stream type fish generally migrate to sea after rearing for at least a year in freshwater. Thus, ocean type fish tend to spend longer periods in ocean habitats compared to stream type populations.

Table 2. A summary of the characteristics of stream and ocean life history types as compiled from various literature sources such as Myers et al. (1998) and Healey (1991).

Stream Type Fish	Ocean Type Fish
Species	
Coho	Coho
Chinook	Chinook
Steelhead	Chum
Sockeye	Pink
Attributes	
Long period of freshwater rearing (>1yr)	Short period of freshwater rearing
Shorter ocean residence	Longer ocean residence
Uses area of ocean north of ocean type	Uses area of ocean south of stream type
Short period of estuarine residence	Longer period of estuarine residence
Larger size at time of estuarine entry	Smaller size at time of estuarine entry
Mostly use deeper, main channel estuarine habitats	Mostly use shallow water estuarine habitats, especially vegetated ones

Information documented in status reviews was used to classify each ESU as either stream type or ocean type according to whether it is comprised primarily of ocean type or stream type populations using information documented in the species status reviews: chinook salmon (Myers et al. 1998), chum salmon (Johnson et al. 1997), and sockeye and steelhead (Busby et al. 1996). We assumed that all populations in aggregate within an ESU fit a general life history type model (Table 2), even though there will be differences between populations within an ESU. Throughout this document, we use the terms ocean type and stream type to distinguish the life history type of an ESU.

As we discussed in the previous section, individual members within a population (or in this case an ESU) employ a variety of alternative spatial and temporal strategies or approaches to using available habitat. We use the phrase **life history strategy** to refer to an approach to the spatial and temporal use of available habitats, including the estuary. To define alternate strategies of estuarine habitat use, we used the size at estuarine entry and the time when they arrive in the estuary. Size at entrance into the estuary can be used to classify life history strategy because there is a linkage between fish size, habitat use, and residence time (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). In general, residence time in the estuary decreases with size at estuarine entry (with the exception of pink salmon). In addition, juvenile salmon are generally distributed along a habitat continuum based upon water depth with the depth of the water occupied by the fish increasing as the size of the fish increases (McCabe 1995). Larger fish can result from growth either in estuarine or freshwater habitats.

The time the fish arrive in the estuary also varies within a general size class of individuals (Carl and Healey 1984; Bottom et al. 2001). Because available resources and habitats can be different depending on when a fish arrives in the estuary, arrival timing represents a reasonable way to define how the fish use habitats. The wide range in variability in size at estuarine entry and time of entry that can occur is illustrated by Figure 2 (reproduced from Bottom et al. 2001). For example, under historic conditions, fry (fish < 60mm at estuarine entry) arrived in the estuary nearly year round while yearlings could be present from February to June. The source populations were not identified in these analyses, but it is highly likely that many ESUs contributed to the pattern of fry appearance historically in the Columbia River estuary.

Based upon size and time of estuarine entry, we defined six general life history strategies that used the estuary historically (Table 3): 1) early fry, 2) late fry, 3) early fingerling, 4) late fingerling, 5) subyearling, and 6) yearling. Fry were defined as fish that enter the estuary at a size < 60 mm with early fry entering in approximately March and April and late fry from May to June. Fingerlings were fish that enter the estuary at a

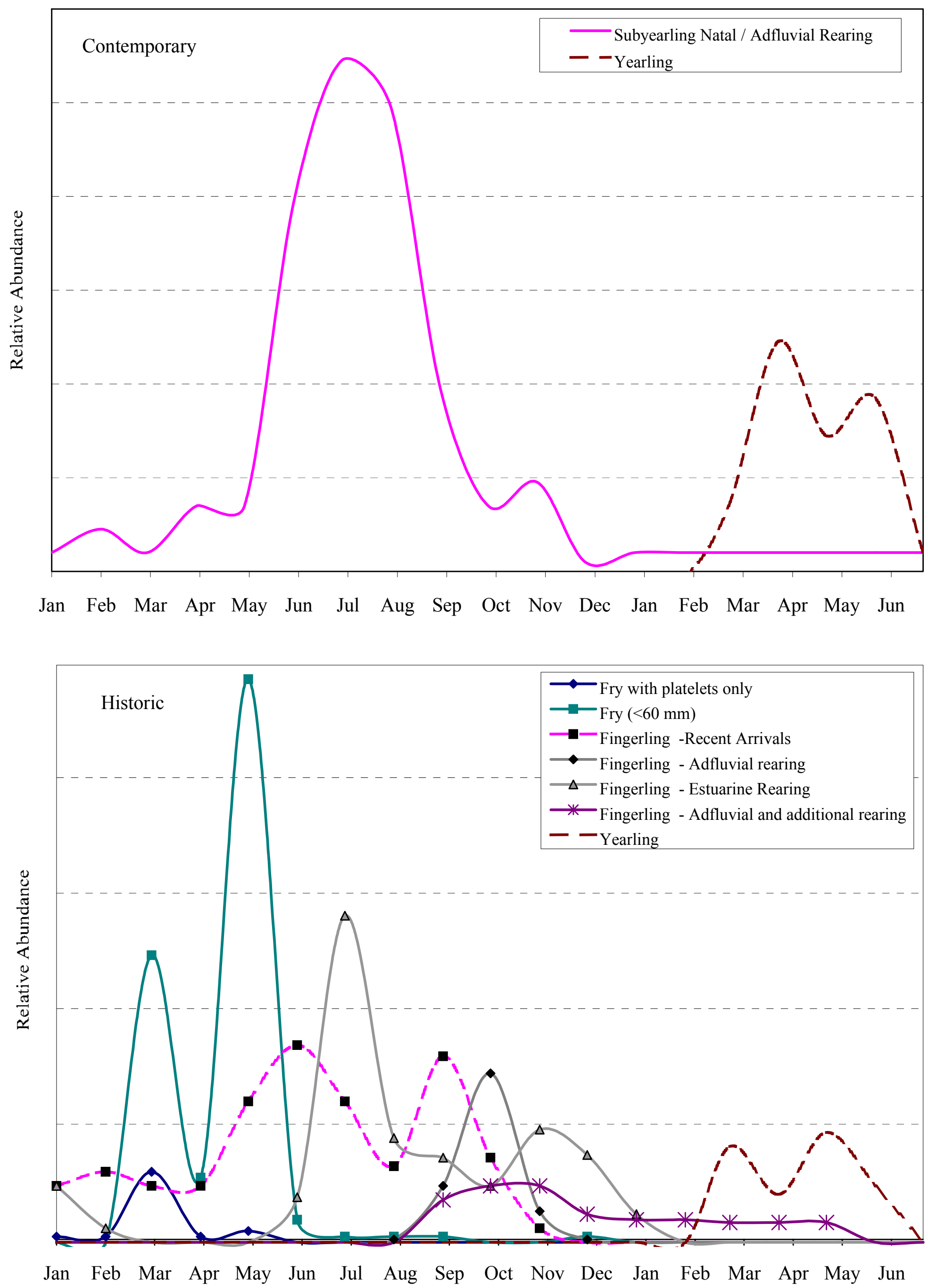


Figure 2. Historical and contemporary early life history types for one brood-year of chinook salmon in the Columbia River estuary. Historical timing and relative abundance based on historical sampling throughout the lower estuary (Rich 1920). Contemporary timing and relative abundance derived from Dawley et al. (1985) sampling at Jones Beach. (From Bottom et al. 2001).

Table 3. Some attributes of life history strategies associated with Columbia River anadromous salmon populations based upon historic use of the system. We used various sources of information, much of which pertains to chinook salmon (e.g., Bottom et al. 2001; J. Burke, University of Washington, pers. comm.), to develop this table. All values should be considered estimates.

Life History Strategy	Attributes
Early fry	Time of estuarine entry: March-April Size at estuarine entry: <50 mm Estuarine residence time: 0-40 days Freshwater rearing: 0-60 days
Late fry	Time of estuarine entry: May-June, present thru Sept. Size at estuarine entry: <60 mm Estuarine residence time- < 50 days Freshwater rearing: 20-60 days
Early fingerling	Time of estuarine entry: April-May Size at estuarine entry: 60-100 mm Estuarine residence time: < 50 days Freshwater rearing: 60-120 days
Late fingerling	Time of estuarine entry: June-Oct, present thru winter Size at estuarine entry: 60-130 mm Estuarine residence time: 0-80 days Freshwater rearing: 50-180 days
Subyearling (smolt)	Time of estuarine entry: April-Oct Size at estuarine entry: 70- 130 mm Estuarine residence time: < 20 days Freshwater rearing: 20-180 days
Yearling	Time of estuarine entry: Feb-May Size at estuarine entry: > 100 mm Estuarine residence time: < 20 days Freshwater rearing: > 1 year

larger size than fry, which implies there was some period of freshwater rearing; fingerlings have yet to begin the physiological transition associated with smolting. Fingerlings rear in the estuary for some period with early fingerlings entering between January and July and late fingerlings from August to December. Subyearlings are fish that rear in freshwater, rear little in the estuary, and smolt as they outmigrate during their first year of life. They may be larger than fingerlings entering the estuary and may reside in the estuary for less time than fry or fingerling salmon. Yearlings rear for at least one year in freshwater and then emigrate; these fish generally spend less time in the estuary than fry, fingerlings, or subyearlings.

We assumed that all populations in aggregate within an life history type/ESU produce a characteristic mix of these strategies when viewed over long time scales, even though there will be differences between populations within an ESU in the relative proportion of each life history strategy that is expressed. We would expect that salmon or steelhead ESUs, which express the stream life history type, are dominated over long time scales by the yearling life history strategy but that other strategies are present in reduced numbers as a way to reduce extinction risk. Similarly, non-yearling strategies would be expected to dominate the characteristic approaches employed by ocean type ESUs. However, yearling strategies would probably be particularly important in those ESUs that are distant from the river mouth.

Considering the habitat changes that have occurred in the estuary, ocean and climate, source populations, and in the freshwater spawning and rearing habitats that produce the source populations, it seems likely that changes in the distribution of life history strategies for each life history type have occurred. This change is suggested by comparing current and historic use of the estuary by different strategies (Figure 2). This figure suggests that all life history strategies were evident for longer periods of time throughout the year (Rich 1919) and the current use of the estuary is more limited now than in the past. While we do not have information on how the distribution of life history strategies changed within a specific ESU or even a life history type, it seems reasonable to assume that there have been significant changes in relative abundance of life history strategies. We extrapolated the likely mix of life history strategies historically employed by each life history type using such information as Figure 2 and the types of changes that have impacted each life history type (Table 4). Comparison of the likely mixes of life history strategies for each ESU under current and historic conditions provides an assessment of the possible change in life history expression by each life history type. Such changes should be considered as hypotheses.

Table 4. Linkage between anadromous salmonid ESU, life history type (ocean or stream type), and dominant life history strategies of juvenile salmon in the Columbia River. We estimated general contribution to the outmigrant population of each life history strategy as Abundant (>50%), Medium (10-50%), Rare (1-9%), or Absent (<1%) listed for each ESU under historic (early 1900s- left side of each cell) and current conditions (right side of each cell). We made these estimates using a variety of data sources and our judgment about how the various changes occurring in the system would have affected each strategy within each ESU.

ESU	Life History Type	Life History Strategy					
		Early Fry	Late Fry	Early Fingerling	Late Fingerling	Subyearling	Yearling
Lower Columbia River Chum Salmon	Ocean	High / High	High / High	Absent / Absent	Absent / Absent	Absent / Absent	Absent / Absent
Snake River Sockeye Salmon	Stream	Absent / Absent	Absent / Absent	Absent / Absent	Absent / Absent	Rare / Rare	Abundant / Abundant
Lower Columbia River Coho Salmon	Stream	Rare / Absent	Rare / Absent	Rare / Absent	Rare / Absent	Rare / Absent	Rare / Absent
Upper Columbia River Steelhead	Stream	Absent / Absent	Absent / Absent	Absent / Absent	Absent / Absent	Rare / Absent	Abundant / Abundant
Snake River Steelhead	Stream	Absent / Absent	Absent / Absent	Absent / Absent	Absent / Absent	Rare / Absent	Abundant / Abundant

Table 4. Continued.

ESU	Life History Type	Life History Strategy					
		Early Fry	Late Fry	Early Fingerling	Late Fingerling	Subyearling	Yearling
Lower Columbia River Steelhead	Stream	Absent / Absent	Absent / Absent	Rare / Absent	Rare / Absent	Medium / Rare	Abundant / Abundant
Middle Columbia River Steelhead	Stream	Absent / Absent	Absent / Absent	Rare / Absent	Rare / Absent	Medium / Rare	Abundant / Abundant
Upper Willamette River Steelhead	Stream	Absent / Absent	Absent / Absent	Absent / Absent	Absent / Absent	Rare / Absent	Abundant / Abundant
Snake River Fall Chinook Salmon	Ocean	Absent / Absent	Absent / Absent	Medium / Rare	Medium / Rare	Medium / Rare	Abundant / Abundant
Upper Willamette Chinook Salmon	Ocean	Rare / Absent	Rare / Absent	Medium / Rare	Medium / Rare	Rare / Medium	Abundant / Abundant
Lower Columbia River Fall Chinook Salmon	Ocean	Medium / Rare	Medium / Rare	Medium / Rare	Medium / Rare	Medium / Abundant	Rare / Rare

Table 4. Continued.

ESU	Life History Type	Life History Strategy					
		Early Fry	Late Fry	Early Fingerling	Late Fingerling	Subyearling	Yearling
Upper Columbia River Spring Chinook Salmon	Stream	Absent / Absent	Absent / Absent	Rare / Absent	Rare / Absent	Rare / Rare	Abundant / Abundant
Snake River Spring/Summer Chinook Salmon	Stream	Absent / Absent	Absent / Absent	Rare / Absent	Rare / Absent	Rare / Rare	Abundant / Abundant

Factors Included in the Analyses

To facilitate recovery of endangered salmon stocks in the Columbia River basin, factors that currently act to suppress their increased viability need to be identified. If the factors are appropriately and correctly pinpointed, it is logical, although perhaps simplistic, to conclude that reducing their affect should improve the recovery potential of targeted populations. Further, incorporating the influence of a factors' impact on recovery of salmon populations should improve policy and management decisions.

Although identifying single factor solutions to salmon survival problems in isolation has not historically been effective, we suggest that identification of limiting factors in the estuary represents a first step that needs to be incorporated into a landscape scale assessment of strategies to improve the recovery potential for endangered salmon populations. Our goal here is to consider the effect of several estuarine factors on recovery of listed anadromous salmonids. We do not wish to imply that estuary changes are the major affect on all ESUs. Assessing the relative role of different limiting factors both within and outside the estuary will require additional analyses and is outside the scope of this review.

The major estuarine related factors that we believe can potentially affect salmonid population viability include climate and climate change (which control other factors), water flow, access to and quality of habitats, sediment, salinity, temperature, toxics, predators (e.g. terns, cormorants, marine mammals, northern pikeminnow), and hatchery and harvest practices. Although it would be useful to evaluate the role of each of these factors, we selected a of these nine factors to analyze based upon whether: 1) a significant change was evident, 2) the factor could potentially affect population viability, and 3) there was quantitative data available that could be used to analyze the affect of the factor within the time we had been allotted.

The factors that satisfied these criteria and were included in this analysis are water flow, availability of salmon habitats, toxics, and predation (primarily Caspian terns). For each of these factors we provide a brief analysis as to how the factor has changed and how the factor could affect population viability based upon existing data and analyses. From these overviews, we developed a series of hypotheses or principles about each factor that helped guide how we rated their relative importance for each ESU.

Analyzing and Rating the Relative Importance of Limiting Factors

To rate the importance of each factor, we developed a simple rating system that ranked each factor as having a high, medium, or low ability to improve the status of anadromous salmon populations. We defined improvement in population status to mean improvement in population viability (McElhaney et al. 2000) as defined by the four VSP performance criteria: abundance, population growth rate, spatial structure, and diversity (McElhany et al. 2000).

We drew inferences about how a factor affects an ESU based upon the life history type of that ESU, how we believed the factor would affect the life history strategies that characterized that life history type, and the hypotheses and principles about each factor that helped we developed from the overviews of each factor. Thus, the limiting factors for all stream type ESUs were ranked similarly while those for ocean type ESUs were ranked similarly. Ratings were developed by considering each factor relative to other estuarine factors within an ESU; ratings were not considered relative to other non-estuarine factors such as tributary habitat.

The rating system consisted of two levels. The level 1 screens evaluated ***if*** the factor was likely a concern for an ESU based upon its affects on VSP and change in that factor from historic conditions. The level 2 screens asked ***how*** the factor affected an ESU based upon where the affects occurred and the life history strategies affected.

Level 1: Is the Factor Likely a Concern for the ESU?

a. **What is the affect on each VSP parameter?** Each factor will have some affect on each VSP parameter. We assumed, however, that if the factor affected large numbers of individuals in the ESU (again relative to other factors) that there was a significant affect on abundance and productivity. Because most populations in threatened or endangered status are at low levels of abundance, we reasoned that these depressed populations needed short term increases in abundance before long term benefits resulting from increased diversity and structure would be useful. Therefore, we doubled the score if the factor affected abundance and productivity. If a factor affected particular life history types or affected specific habitat types more than others, we assumed there was an impact on spatial structure and diversity.

b. Has the factor changed from historic conditions and what it is the likelihood that it could be improved relative to the other factors? We considered whether each factor had changed significantly from historic conditions. Because we intentionally selected factors that we believed had changed significantly from historic conditions, this screen did not differentiate much between factors. We also considered from a practical perspective how much change in each factor was possible since a factor could be significantly changed from historic levels but relatively difficult to change relative to other estuarine factors.

Level 2: How Does the Factor Affect the ESU?

a. Does the factor have a significant affect on the abundance of the dominant life history strategy? For the dominant life history strategy, we asked how the factor affected the abundance of juveniles of that life history type in shallow water habitats (within the confined portion of the estuary), deep water habitats (within the confined portion of the estuary) and plume habitats. Although there are multiple zones and numerous habitat types within each zone, knowledge of how different juvenile life history strategies specifically use these habitats and zones is limited. Thus, we could not differentiate effects of limiting factors on different habitat types (e.g., emergent marsh vs swamp vs mudflat) within a zone and between zones (e.g., river mouth vs riverine tidal).

However, available information does adequately demonstrate that there is differential use of shallow and deep water by salmon juveniles within the confined portion of the estuary (Bonneville Dam to the river mouth) based upon their life history strategy. Fry and fingerling strategies are more closely associated with shallow, low velocity habitats (e.g., swamps, emergent marshes, and shallow flats) and less associated with medium and deep, higher velocity channel habitats in the analysis; the opposite pattern exists for larger size classes such as yearling (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). Thus, we collapsed the estuary from Bonneville Dam to the mouth into one zone and the plume was considered a second major zone. Within the Bonneville to the mouth zone, we also differentiated shallow, low velocity habitats from medium and deep, higher velocity channel habitats.

b. For the dominant life history strategy, does the factor affect habitat quality, quantity, and opportunity? For the dominant life history strategy, we asked what type of affect the factor had in shallow water, deep water and plume habitats. We considered effects of the factor on habitat quantity, quality, and opportunity. The concepts of opportunity and quality (or capacity) metrics were proposed by Simenstad and Cordell (2000) and adopted by Bottom et al. (2001) for the Columbia River estuary.

Opportunity attributes relate to the accessibility of habitat to juvenile salmon. In general, opportunity metrics are largely physical and chemical in nature such as tidal elevation, temperature, and location of habitat. For example, extreme high temperatures and diminished flows can constrain accessibility of shallow water habitat. Capacity measures primarily relate to the biotic and ecological functions (i.e., acquiring food and avoiding being eaten) of habitat. Capacity metrics must be considered within the context of the species and life stage using the habitat and the location of that habitat within the landscape. In addition to capacity and opportunity, we also included quantity of habitat as a separate metric. For toxics, we rated affects separately in shallow water and deep water estuarine habitat for water borne and sediment borne contaminants. For example, if there were risks to the main life history type from both types of contaminants in shallow water, then the score would double.

Each of the four questions listed above was evaluated for each factor for each ESU based upon whether they were classified as an ocean or stream life history type ESU. Each cell in a matrix was either scored as a yes (+1) or no (0) with two exceptions: 1) abundance and productivity which were given a +2 score, and 2) toxics in deep and shallow water which each could be scored a +2 if there was effects from both water borne and sediment associated toxics. This affected the maximum possible score that could be assigned to a factor. For flow, habitat and predation, the maximum possible score was 20 whereas the maximum possible toxic score was 28. The final rating was computed as the ratio between the assigned score and maximum possible score; a ratio of >0.67 was assigned a high ranking, 0.34-0.66 a medium ranking, and <0.33 a low ranking

FACTORS LIMITING SALMON USE OF THE COLUMBIA RIVER ESTUARY

To facilitate recovery of endangered salmon stocks in the Columbia River basin, it is important to identify factors that currently act to suppress their increased viability. Here we review, the role of water flow, availability of specific estuarine habitats, toxics affecting the quality of habitats, and predation by Caspian terns.

Water Flow

Water is the very essence and foundation for salmon use of any area. Water clearly allows the connection between the terrestrial oriented land mass (the freshwater zone) to the submerged land masses dominating the globe (the marine zone), fulfilling a major role of the estuary, a migratory corridor connecting these two principal zones. The freshwater, estuary, and marine zone represents the full landscape spectrum salmon occupy. However, more importantly, water interacting with land forms the habitat that salmon occupy. Variation in amount of water in relation to where it interacts with land provides the diversity of habitats that salmon utilize to complete their life cycle.

The estuarine features to which salmon life histories have adapted are largely the result of water associated riverine and tidal processes that transport various materials such as sediments, biota; establish salinity and temperature gradients; and regulate water levels and velocities. The highly productive nature of "pulsed" estuarine systems is a direct result of this dynamic interplay between river and tide. For this reason, the role of water flow in creating and shaping estuarine habitat in the Columbia River is clearly an important attribute to consider in salmon recovery in the basin.

The shaping of estuarine habitats is, however, not controlled entirely within the estuary, but is determined by regional and basin-wide variations in climate affecting hydrology and ocean conditions. Because these "external" factors establish the physical template for the entire estuary, they also directly or indirectly affect each of the major attributes of salmonid performance, availability of estuarine habitat (habitat opportunity), the quality of estuarine habitat (habitat capacity), and salmon population structure and life history.

We start with the terrestrial backdrop influencing the delivery of water to the estuarine zone, i.e. the watershed. In general terms, the Columbia River has the largest average flow (~7,300 m^3s^{-1}) of any river on the Pacific coast of North America (Church and McLean 1992, Jay and Naik 2000). The Columbia River contributes some 60% (winter) to 90% (summer) of the total freshwater input into the Pacific Ocean between

San Francisco and the Strait of Juan de Fuca and strongly affects regional seawater properties of the Northeast Pacific Ocean (Barnes et al. 1972).

The major geologic feature affecting flows through the basin is the Cascade mountain range. The Cascades of Oregon and Washington divide the Columbia River drainage basin into interior and western sub-basins. The moist and relatively warm western sub-basin contains only ~8% of the total surface area of the 660,480-km^2 basin, but contributes almost one-quarter of the total river flow (Orem 1968). Most of the western sub-basin is at too low an elevation to accumulate a large seasonal snow pack. Thus, the highest flows are observed during and shortly after winter storms, between December and March. In contrast, most of the flow in the interior sub-basin occurs as the result of melting of a seasonal snow pack between April and June. Much of the interior sub-basin is relatively arid, but its Canadian component experiences heavy winter snowfall and plays a major role in spring freshet flows.

Climate is another element that clearly affects the amount of water delivered to the estuary and the connection between the freshwater and marine environments. Climate-induced variations in Columbia River flow occur on time scales from months to centuries (Chatters and Hoover 1986, 1992). A singular measure of climate shown to relate to salmon survival is the Pacific Decadel Oscillation, commonly know as the PDO (Mantua et al. 1997). The PDO has been shown to cycle in approximately 30-year time scales, alternating between a cold and warm phase. During the cold phase, more rainfall is typical in the Pacific Northwest, whereas in the warm phase, less rainfall occurs. The PDO primarily reflects sea surface water temperatures in the North Pacific Ocean and is associated with the positioning of the Aleutian Low Pressure Zone over the North Pacific (**Beamish et al ???**).

The cold phase of the PDO (e.g., the 1945-1976 period) was regarded as benefiting salmonid production in the Pacific Northwest while being less favorable for salmon originating in northern British Columbia and Alaska. The opposite circumstance prevails during the warm phase, characteristic of the recent period between 1977 to about 1998, when listing of salmon in the basin occurred. Another cold, wet phase seems to have commenced about 1998 (Peterson and Schwing 2003). These PDO-related fluctuations in salmonid survival have been linked to the degree of density stratification of the coastal ocean (Gargett 1997), but they are also likely influenced by conditions within the river and estuary (e.g., salinity, turbidity, and river flow itself).

Another climate-related feature known to influence weather and conditions in the Pacific Northwest is the phenomena associated with the El Niño-Southern Oscillation (ENSO; typically 3-7 years in duration) index (Redmond and Koch 1991). ENSO cycles are shorter in duration compared to the PDO cycle, typically amplifying conditions associated with the cold or warm phase of the PDO. El Niño winters in the Pacific Northwest often bring high sea level, warm air temperature, low precipitation, low snow-pack, and weak subsequent spring freshet flows (Kathya and Dracup 1993; Dracup and Kathya 1994). La Niña winters (the contrast to El Niño) typically exhibit an opposing climate and hydrological response. As a consequence, the annual average flows of the Columbia and the Willamette Rivers during years with a strong El Niño winter are 91 and 92% of the long-term annual average, while in case of strong La Niña winters, they are 110 and 111%, respectively. Considering PDO cycles alone, average annual Columbia River flows at The Dalles and Willamette River flows, respectively, were 109 and 102% of the long-term average in the 1890-1921 cold phase, 86 and 87% of average in the 1922-1944 warm phase, 102 and 110% of average in the 1945-1976 cold phase, and 88 and 88% of average in the 1977-1995 warm phase (Jay 2001).

El Niño effects are intensified during a warm-PDO phase, while those of La Niña are enhanced during a cold-PDO phase (Gershunov et al. 1999). The net effect is that during an El Niño/warm-PDO combination, respective average annual Columbia River flows at The Dalles and Willamette River flows are 85 and 81% of the long-term annual average, while in case of a La Niña/cold-PDO combination, they are 111 and 119%, respectively (Figure 3). (These differences are significant at the 95% confidence limit.) Conversely, El Niño effects are suppressed during the cold-PDO phase as are those of La Niña during the warm-PDO phase (Jay 2001).

The Columbia Basin's climate response is conditioned by its position between 41°30' and 54°40'N lat., within a latitudinal band of strong response to the ENSO cycle and to the PDO (Mantua et al. 1997). While the flow per unit area is much larger in the western than in the interior sub-basin, there are only modest variations across the basin in response to ENSO or PDO forcing. Still, the relatively large north-south extent of the basin brings about important differences in flow seasonality--the incidence of winter floods and timing of spring snowmelt--even within the interior sub-basin.

Thus, climate introduces a wide degree of variation in water distributed throughout the basin and watershed, obviously affecting water flow. This is critical in that natural variations in Columbia River flows associated with El Nino and the PDO affect habitat conditions in the estuary. This helps determine what areas are wetted and potentially accessible to juvenile salmon, changes estuarine salinity gradients, influences

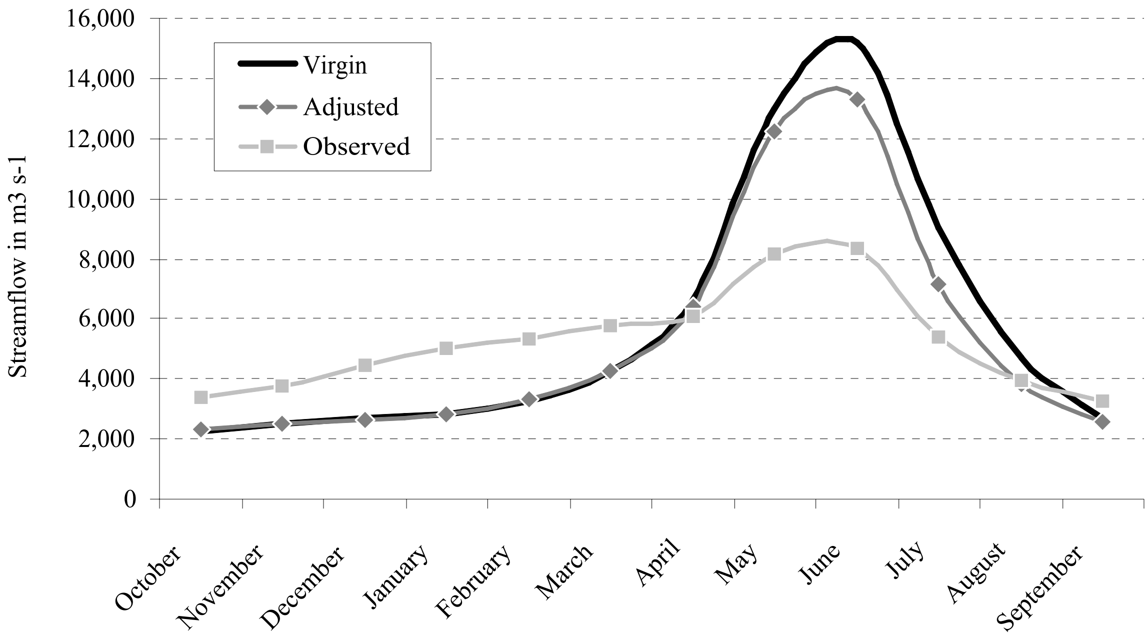


Figure 3. Comparison of the monthly averaged Columbia River interior sub-basin virgin, adjusted, and observed river-flow estimates 1970-1999. Flow regulation and irrigation depletion have greatly decreased spring and summer flows (May to August), while increasing flows from September to March. (From Bottom et al. 2001).

sediment transport processes, and alters the distributions of marine and freshwater species with which salmon interact. It is against this backdrop of varying climatic conditions that we consider the role of water flow and its effect on habitat important to salmon in the Columbia River estuary.

Changes in annual average flow and when water arrives in the estuary are an integral measure of changes in a river system. In a recent analysis and review, Jay (as reported in Bottom et al 2001) provided a detailed estimate of changes in flows in the Columbia River for the past 100 years, thereby providing an accounting of flow conditions during the historical and current period.With respect to overall water delivery through the estuary, Jay concluded that there has been approximately a 16.4% reduction in flow during this period. Moreover, he made an assessment of the contribution of climate and human perturbation on the observed flow and concluded that approximately half of the change was due to climate (less rainfall) and half to human activities (water withdrawal for irrigation). In addition, a small percentage of the decrease was assigned to a combination of uncertainty (error) and evaporation from the impoundment of water in reservoirs due to increased surface area in the basin. Jay's conclusions were based upon the record of observed flows for the past 100 years and an estimated adjusted flow provided by the USGS to account for reservoir manipulations; this was used estimate the virgin river flow (flow unadulterated due to hydropower operations and irrigation removal) for the past 100 years (Figure 3).

The above changes in annual average flow are only a small part of the total hydrological changes in the Columbia River basin. Seasonal changes, particularly those in spring freshet timing and magnitude, have been much larger than those in annual average flow. Spring freshets are extremely important for juvenile salmonids in that high flows (especially overbank flows) provide habitat, limit predation by increasing turbidity, and maintain favorable water temperatures during the spring and early summer. Organic matter supplied by the river during the freshet season is also a major factor maintaining a detritus-based food web, centered in the estuarine turbidity maximum (ETM).

Very large freshets before modern flow regulation (i.e., before ~1970) lasted 30 to 60 days, with the sharpness of the peak largely governed by the relative timing of snowmelt throughout the basin. Flows in the Columbia River interior sub-basin (the flow measured at the The Dalles) are primarily driven by spring snowmelt, although there are rain-on-snow freshets in some winters.

Jay (2001) provided some critical analyses that documented the magnitude and timing of the freshet and subsequent change. Before 1900, the highest flows typically occurred during May-July. Monthly Columbia River virgin flows at The Dalles were 11,480 m^3s^{-1} (for May), 16,760 m^3s^{-1} (for June), and 12,600 m^3s^{-1} (for July) during 1879-

1899. The corresponding figures for 1945-1999 were 13,300 m^3s^{-1}, 15,840 m^3s^{-1}, and 9,420 m^3s^{-1}; these values represent changes of +15.9, -9.5, and -25.2%, respectively.

These flow changes were amplified throughout the basin leading Jay (2001) to conclude that flow regulation decreased spring freshet magnitude and increased flows during the rest of year through winter draw-down of reservoirs, filling of the reservoirs during the freshet, and de-synchronization of flow peaks throughout the basin (Figure 4).

More specifically, Jay apportioned the timing and magnitude of the freshet change to climate change, water withdrawal and flow regulation. Jay found the flow decrease in the freshet period resulting from climate was 5.6%. Furthermore, he noted that the January-July virgin flow average for 1879-1899 was 8,050 m^3s^{-1}, while for 1945-1989 it was 7,850 m^3s^{-1}, a decrease of only 2.5%. Thus, most of the loss of freshet flow represents flow that now occurs during winter, early spring, or late summer and fall.

Similarly, the present decrease in freshet season flow due to water withdrawal was an estimated 10.5% (a reduction of 5.7% for May, 12.5% for June, and 20.8% for July, respectively. Finally, the present freshet season flow decrease due to flow regulation was overall estimated to be 33.1% (a reduction of 31.6% for May, 32.4% for June, and 19.8% for July, respectively. Flow regulation is clearly the source of the largest reduction in spring flow. The total reduction in freshet season (May-July) mean flow due to climate change, irrigation depletion, and flow regulation is 5,870 m^3s^{-1} or 43% of the virgin flow for this period.

The timing of the maximum spring freshet flow has also changed (Figure 5). Maximum daily spring freshet flow now typically occurs at about water-year Day 242 (29 May), whereas maximum flow occurred in the 19th century at about water-year Day 256 (12 June), a change of about two weeks. In terms of the phase of the annual flow fluctuation, the freshet is about a month earlier. Part of this change is due to climate warming, but a component is also due to pre-release of water for flood control before the spring freshet. Irrigation withdrawal usually peaks in June, which tends to further curtail the freshet.

Another feature of water flow significant to salmon is the occurrence of overbank flows. The historic bankfull flow level is estimated at about 18,000 m^3s^{-1} for the main stem below Vancouver (Jay 2001). Modern bankfull level is set by the standard project flood level of ~24,000 m^3s^{-1} for the lower river. According to Jay (2001) some overbank flow occurred in many years before 1900, both in winter and in spring (Figure 6) whereas substantial overbank flow (above 24,000 m^3s^{-1}) is now rare, with significant events

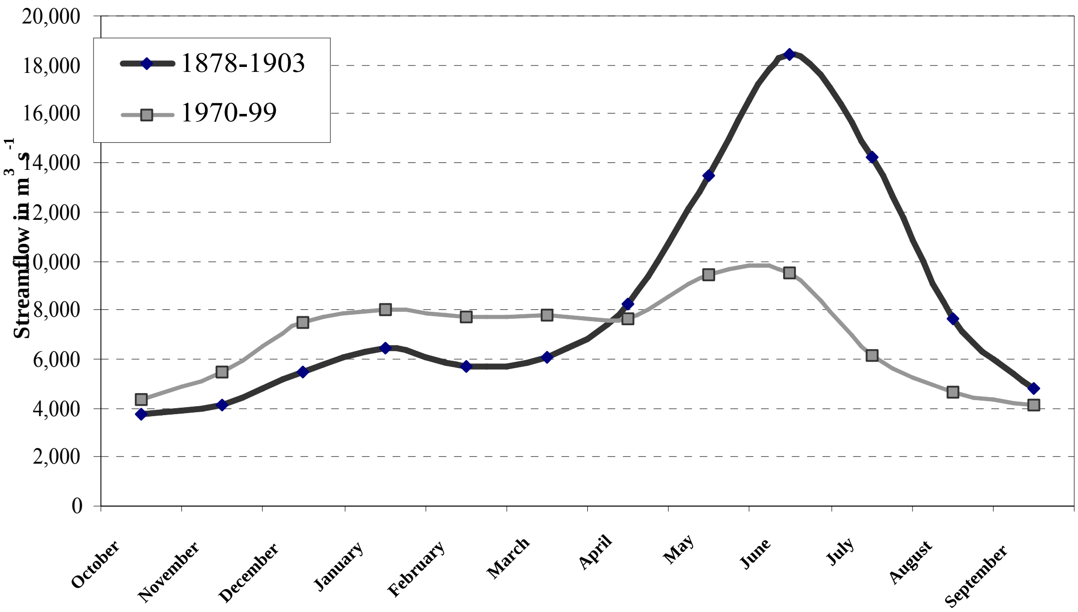


Figure 4. Changes in the annual flow cycle of Columbia River flow at Beaver, 1878-1903 vs. 1970-1999. (From Bottom et al. 2001).

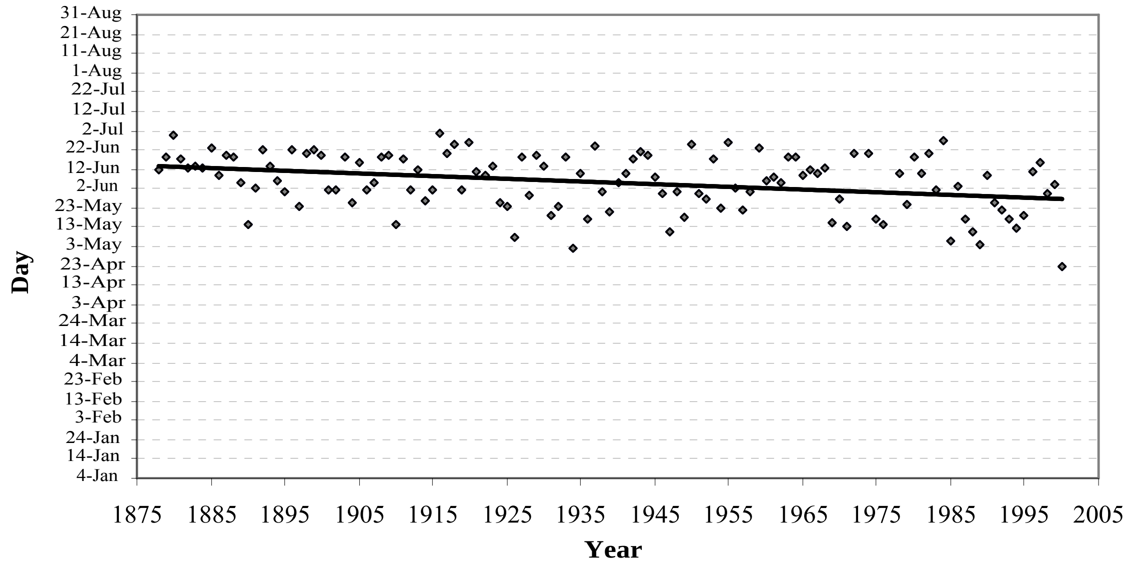


Figure 5. Peak freshet day vs. year suggests that the freshet is now about two weeks earlier than in the 19th century in the Columbia Basin. (From Bottom et al. 2001).

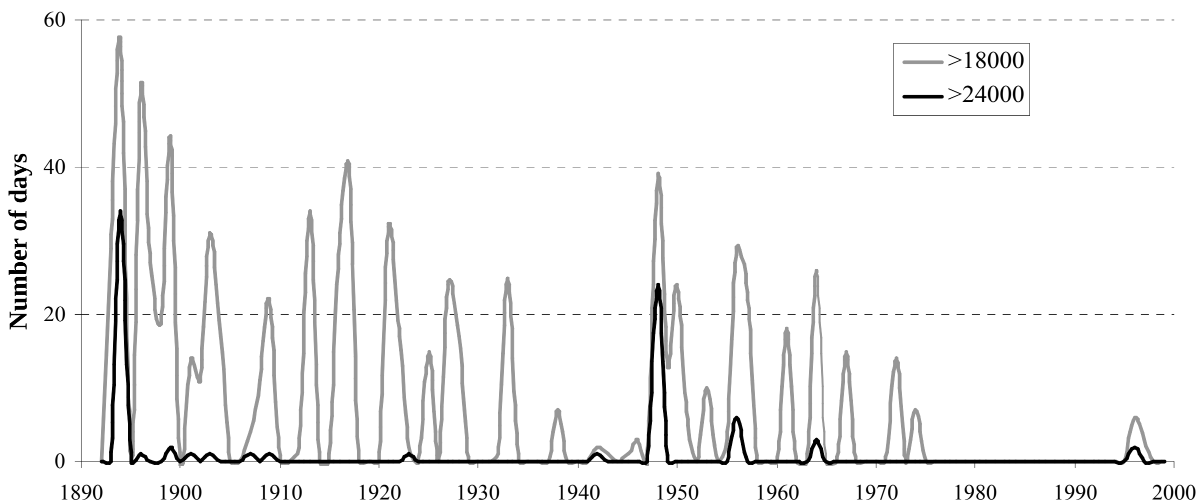


Figure 6. The incidence of flows above 18,000 m^3s^{-1} (the pre-1900 estimated bankfull flow level) and above 24,000 m^3s^{-1} (the present bankfull flow level). The present bankfull flow level has only been exceeded in four years since 1948. (From Bottom et al. 2001).

occurring only five times during the last half century. Historical bankfull levels of 18,000 m^3s^{-1} are rarely exceeded due to effects of flood control measures and irrigation depletion. The season when overbank flow typically occurs has also shifted from spring to winter, because western sub-basin winter floods (not interior sub-basin spring freshets) are now the major source of such flows (Jay 2001). Climate was found to be a secondary factor in the incidence of overbank flow (Jay 2001). Overbank flow events were clearly more common during the cold-PDO phase (1945-1977) than during the preceding warm-PDO phase (1921-1944), even though the degree of flow regulation and irrigation depletion grew over time (Figure 7). Nevertheless, Jay concluded flood protection, diking, flow regulation, and water withdrawal largely eliminated climate influence on overbank flow. Overbank flow was found to be rare during the more recent cold-PDO phases, and was totally absent during the last PDO warm phase (1977-1995).

The effect of flow changes is not restricted to the area traditionally considered the estuary in the Columbia River system. NOAA Fisheries has been investigating the role of the Columbia River plume as habitat important for salmon and steelhead since 1998. The primary issue being addressed is whether anthropogenic modification of the plume habitat has affected recovery of salmon populations from the Columbia River basin.

Incumbent on this assessment is development of empirical evidence that the plume has a role that influences how juveniles make the transition from a freshwater to marine environment. The evidence to date suggests the plume serves salmon in multiple ways. For example, the plume appears to facilitate primary production during the spring freshet period. During low flow years, such as observed in 2001, the amount of chlorophyll evident off of the Oregon and Washington coast affiliated with the plume, as characterized with satellite observations by SeaWifs, was much lower than observed when more normal flows occurred, such as observed in 1999 (Thomas et al. 2003).

The plume also serves to distribute juvenile salmon in the coastal environment. In May and June when flows are higher, juveniles are found further offshore, in the low saline waters they appear to prefer, than when flows are lower (Figure 8). During the years when less flow out of the Columbia River is evident during the freshet period, salmon are more localized around the mouth of the Columbia River. Pearcy (1992) hypothesized that one function of the plume was to distribute juvenile salmon offshore, away from predation pressure closer to the shoreline. Our findings are consistent with this proposed role. Another feature of the interaction of the freshwater and saltwater characteristic of the plume is frontal features which are thought to concentrate food resources important for juvenile salmon. Findings to date support this function.

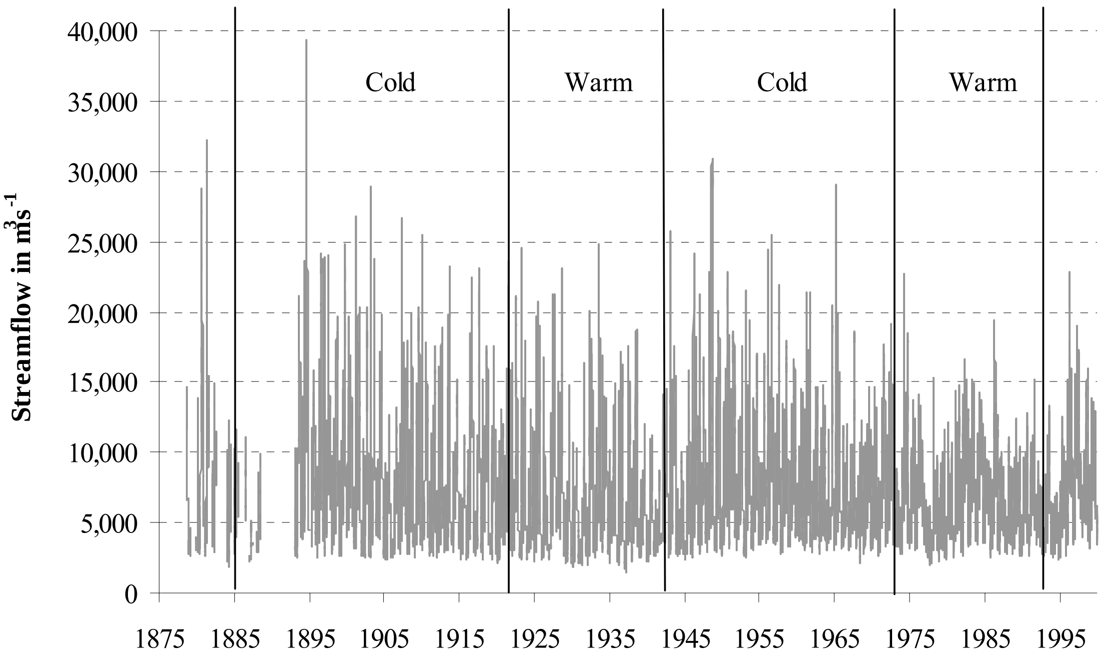


Figure 7. Monthly average flows at Beaver (1878-1999), present and historical bankfull flow levels, and warm and cold-PDO cycles. Historically, there was a major difference between the warm and cold phases of the PDO cycle in disturbance frequency. This has been largely eliminated by flow regulation and diking; overbank flow is now a rare event. (From Bottom et al. 2001).

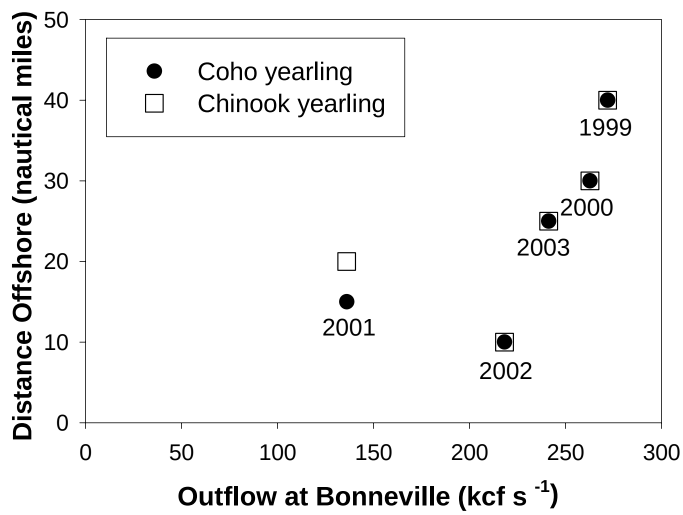


Figure 8. Relationship between average river flow registered at Bonneville Dam for 10 days prior to sampling in the Columbia River plume in May and the maximum distance offshore juvenile chinook and coho salmon were captured in surface trawls along a transect extending east along latitude 46.15, just south of the mouth of the Columbia River.

Zooplankton biomass is in fact highly associated with frontal features at the plume margins (Figure 9) and less so either in the plume or oceanic zones. Our supposition that juvenile salmon would preferentially utilize frontal features was, however, not validated. Juvenile salmon were not higher in abundance exclusively around frontal features. We did conclude that smaller juvenile salmon showed a significant preference for the plume and front habitats as compared to the more marine, oceanic habitats (Figure 10). NOAA Fisheries is developing evidence that salmon continue their preference for the low saline environment of the plume, as they retain their orientation to the surface region (Emmett et al. 2003). The higher turbidity associated with the low salinity plume waters is considered to provide refugia from predators.

A result of developing empirical linkages for a role for the plume to facilitate growth and survival of juvenile salmon is focusing on attributes of the plume that can be used to define habitat important to juvenile salmon. Features such as the surface area of the plume, the volume of the plume waters, the extent and intensity of frontal features, and the extent and distance offshore of plume waters are now considered surrogate physical attributes defining habitat important to salmon. Obviously, flow from the Columbia River can modulate the features characterized as defining habitat in this dynamic estuarine zone. Factors, both natural and anthropogenic that modify flows can logically be considered to modify habitat used by salmon as they make the transition to a marine life. In this context, lowering flows beyond what would normally occur would be considered to diminish the availability of these habitats in the plume region for salmon.

Evaluating the impact of water flow on habitat utilized by salmon is a challenge, but remains the crux of the analysis needed to put flow changes in the basin into perspective. Several recent analyses provide some empirical evidence on the role of the altered flow regime described above on habitat in the Columbia River estuary. Baptista (2001) using a hydrologic model developed specifically for the Columbia River found the estuary during the historic period (late 1800s) was able to sustain habitat features defined to be important to salmon (characterized as water velocities less than 30 cm/sec--important to smaller juvenile salmon) to a greater degree in the face of ever increasing water flows than is evident now (Figure 11).

Physical changes in the lower Columbia River estuary (discussed in the next section) and the loss of resilience in maintaining estuarine habitat attributes important to salmon has altered the amount and timing of water delivery to the estuary. These changes will significantly affect availability of habitat needed in the estuary to sustain the diverse life history strategies for the various source population of salmon and steelhead.

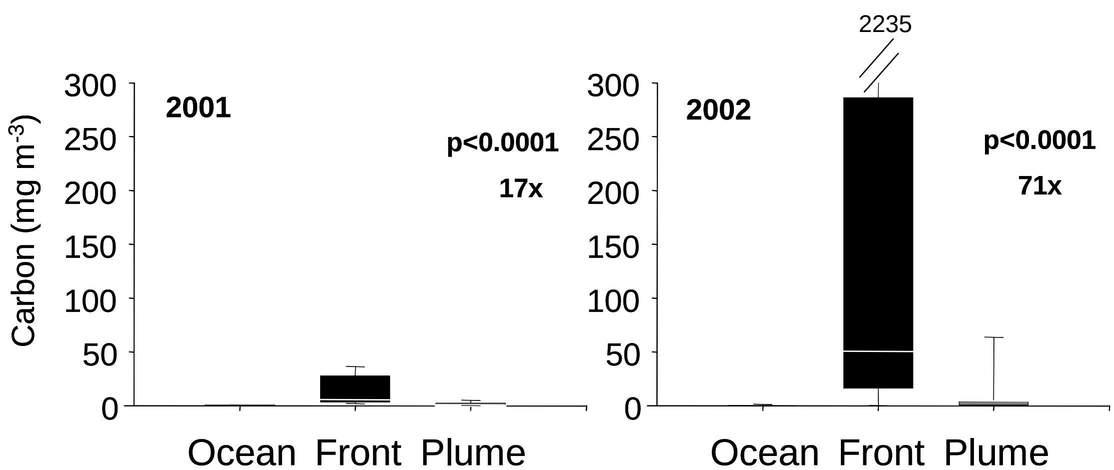


Figure 9. Biomass of *Cancer magister* megalopae captured in May 2001 and 2002 in the ocean, front and plume habitats using a neuston net. Box plots demarcate the 10^{th}, 25^{th}, 50^{th}, 75^{th} and 90^{th} percentiles. An ANOVA (blocked) was used to identify significant differences. Biomass of this species was 17 and 71 times higher in the front habitat compared to the average of the ocean and plume habitats.

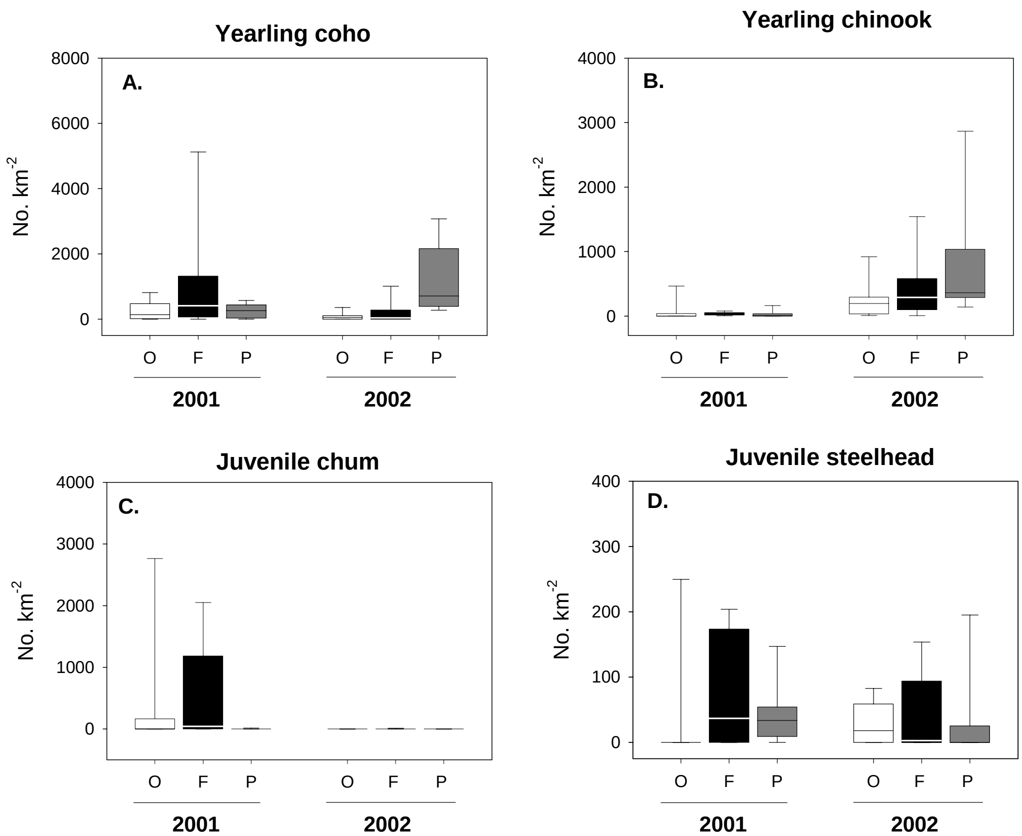


Figure 10. Abundance of A) yearling coho B) yearling chinook C) juvenile chum salmon and D) juvenile steelhead captured in the ocean, front and plume habitats using a Nordic rope trawl. Box plots demarcate the 10th, 25th, 50th, 75th and 90th percentiles of densities. From Robertis et al. (submitted) Frontal regions of the Columbia River plume: II Distribution and feeding ecology of juvenile salmon. Can. J. Fish. Aquat. Sci.

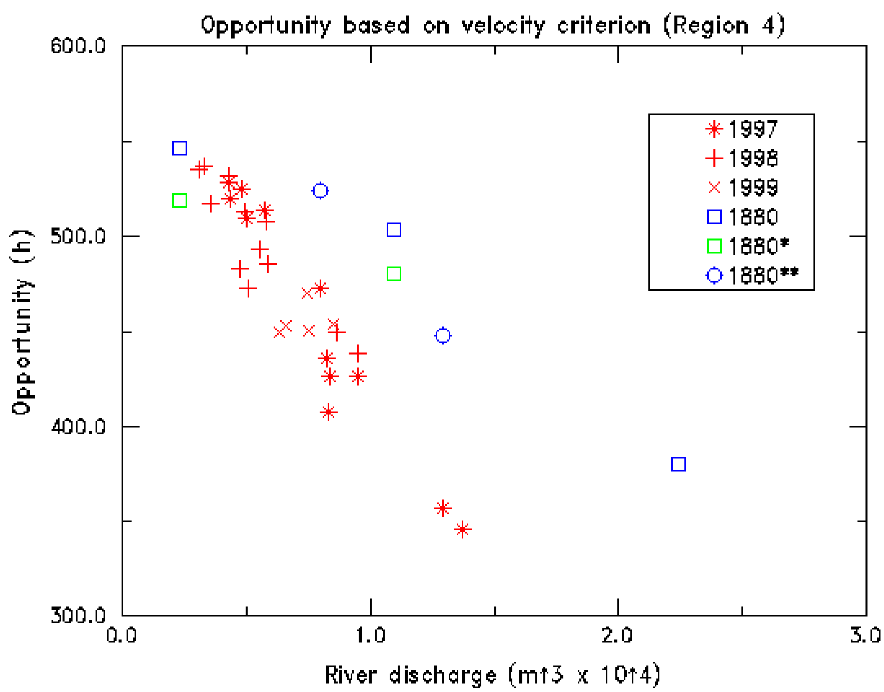


Figure 11. Habitat opportunity in Cathlamet Bay as a function of river discharge. Habitat opportunity is defined as a site with water velocity less than 30 cm/sec. Habitat opportunity shown in hours normalized to a 30-day month is based on this velocity criterion. There are marked differences between habitat opportunity in the modern and pre-development systems. (From Bottom et al. 2001).

Another recent analysis reveals the loss of habitat considered important to juvenile salmon that can be linked to reduced flow. The analysis is especially important because it considers the role of reduced flow on habitat in the tidal freshwater zone of the Columbia River estuarine system, an area seldom evaluated. Kukulka and Jay (2003) demonstrated that there was approximately a 62% loss of shallow water habitat (defined by depth between 10 cm and 2 m) that was attributable to diking (physically removing access of water to the tidal floodplains) and the reduction of peak flows by 40% (consistent with freshet flow reduction discussed earlier) for the region between river mile 50 to river mile 90 on the Columbia River (Figure 12). The analysis incorporated the spring freshet period, when maximal use of estuarine habitat by different life history types employing a variety of strategies appears to occur (NOAA Fisheries, unpublished data). Diking and flow reductions have reduced shallow water habitat in the freshwater tidally influenced region of the Columbia River estuary by 52% and 29%, respectively.

The diverse life histories of juvenile salmon are an adaptive response to longstanding features of the physical and biotic environments they experience. Diversity of salmonid rearing and migration behaviors thus are linked to various habitats and environmental conditions that can support each developmental stage (e.g., egg, fry, smolt, etc.). Within a suitable range of times and locations, individuals can therefore fully complete their life cycles and maintain membership in a population. Major departures from the historical template of an ecosystem can thus potentially create mismatches between established salmon behaviors and the physical environment or, similarly, prevent the expression of potential behaviors by eliminating habitat opportunity. The hydrological changes described above, particularly those associated with flow regulation and water withdrawals, along with floodplain diking discussed in the next section, represent a fundamental shift in the physical state of the Columbia River ecosystem. Such changes may have significant consequences for both salmonid diversity and production.

The effort to stabilize flows in the Columbia River basin ironically may create less stable conditions for salmonids whose migration and rearing behaviors have adapted to historical patterns of hydrologic variability. Of particular importance is the reduction of the spring freshet to which the timing of downstream migrations and patterns of habitat use of some subyearling and yearling life-history types may have been linked. One potential result of dampening flow variations in the Columbia River could be a greater uniformity of migration patterns with potential consequences in the timing and sizes of salmon arrival in the estuary and/or ocean.

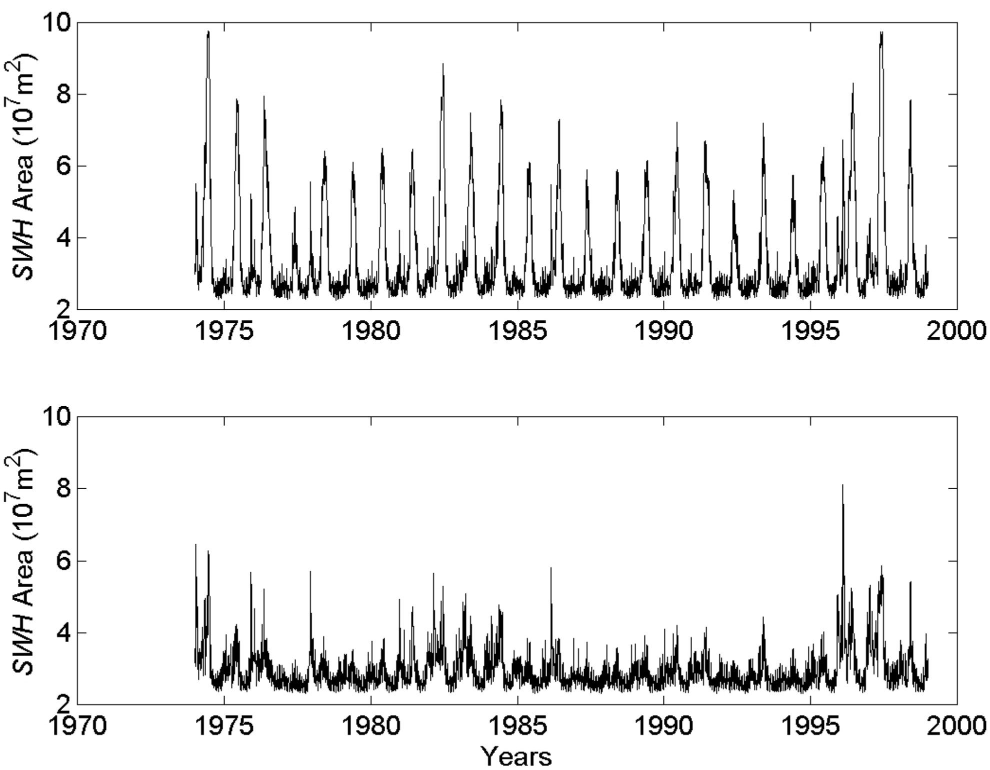


Figure 12. The change in availability of shallow water habitat in the tidally influenced region between RM 50 and RM 90 on the Columbia River under unmodified and modified flow conditions only. The top panel represents condition under virgin flow with no dikes, where extensive inundation of the floodplain occurs for long durations. The bottom panel represents conditions under modern flow conditions with no dikes, where river staged lowered and much less inundation of floodplain for shorter duration occurred. (From Kukulka and Jay 2003).

The nearly complete elimination of overbank flooding throughout the expansive tidal freshwater portion of the estuary may pose some of the most significant consequences for Columbia River salmonids. Flow regulation and diking effects together have largely eliminated access to off-channel floodplain habitats and refugia during high flow events. If, as we suspect, patterns of extended estuary use by small subyearling migrants are directly linked to the availability of shallow-water habitat, the loss of the tidal floodplain could simplify salmon diversity and reduce rearing capacities of estuarine habitats.

In addition to the physical effects of reduced habitat opportunity on salmon diversity, flow regulation, in conjunction with floodplain diking, may influence the productive capacity of the estuary by regulating so-called "energetic processes" such as food production, competition, and predation. Floodplain inundation greatly increases the surface area of tidal estuarine and riverine habitats available to salmonids, allowing fish to expand their distribution into productive off-channel areas and may relax competitive interactions by reducing fish densities.

For example, recent studies in a non-tidal portion of the lower Sacramento River found that tagged juvenile chinook salmon released in the seasonally-inundated floodplain had better growth, higher consumption rates, and improved survival compared with others released into the main river channel (Sommer et al. 2001). Elimination of overbank flooding also prevents the pulsed delivery of structural and energetic components to the rest of the estuary, including large wood, sediments, detritus, and prey organisms produced in adjacent riparian and floodplain habitats.

In summary, flow is a fundamental factor affecting characteristics of salmon and their habitat in the estuary and plume. Large scale effects on flow occur as a result of spatially explicit interactions of short and long term climate cycles (ENSO and PDO, respectively) with the watershed. The generation of electricity, flood control, and irrigation have had significant affects on attributes of flow. These include a reduction in the mean annual flow, reductions in the size of the spring freshets, an almost complete loss of overbank flows, and changes in timing of ecologically important flow events. The hydrological changes, along with floodplain diking, represent a fundamental shift in the physical state of the Columbia River ecosystem.

Such changes potentially have significant consequences for both expression of salmonid diversity and productivity of the populations by affecting quality of habitat available, its accessibility and quantity. In particular, because the changes in habitat are most pronounced in shallow water areas, effects on the ESUs and life history strategies (the fry and fingerling strategies) that use these and depend upon these shallow water areas is most significant. We conclude that flow alterations are a significant limiting factor in the plume. Primarily as a result of flow, but also no doubt as a result of physical changes to the estuary (eg., dredging and diking), the shape, behavior, size, and composition of the plume has been changed.

Habitat

The role of habitat has been a defining and consistent objective in many studies focusing on salmon biology, particularly in the freshwater areas. Characterizing habitat and its role is critical in assessing needs of salmon using spawning and rearing habitat in the tributary and mainstem rivers. Defining habitat in these cases has largely been successful because the juveniles and adults are largely fixed in space for relatively long periods of time. However, once the juveniles begin moving from mainstem and tributary rearing habitats, defining habitat attributes using empirically defined specific criteria has been elusive. Although it is unclear why this is the case, the concept that salmon during migratory phases are using habitats simply as corridors to pass from one area to another has probably narrowed our expectations of how habitat functions during this phase.

With respect to the estuary, habitat is important, but exactly what constitutes salmon associated habitat in the Columbia River estuary is unclear. The need for more quantitative descriptions of habitat attributes important to salmon has intensified recently as the calls for habitat restoration actions have increased in response to mandates to reduce risks to endangered salmon populations. The recovery of 'what' represents the most common response to this call. In lieu of specific knowledge, restoring the entire estuary for all plants and animals becomes the primary option embraced (LCREP 1999). This type of broad scale approach is limiting, however, because estuaries are typically extensively urbanized environments and so any changes that restore historical attributes are costly, particularly those associated with land acquisition. Questions quickly erupt as to what specifically needs to be restored, once real costs for the broad brush approach become evident.

Although specific studies in the Columbia River estuary are limited, research in estuarine systems has demonstrated that estuaries are composed of a variety of habitats that are used differently by different salmon types that can be distinguished by their life history strategy (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). One of the defining features of life history strategies employed by salmon that affects habitat specific use is juvenile size. Larger juveniles typically use deeper water habitats within the estuary, located more centrally to mainstem channels whereas smaller juvenile salmon use the more peripheral side channel areas associated with the more shallow water habitats (McCabe 1986). Large and small, in these examples, are consistent with salmon that exhibit the yearling life history and to some extent, for juveniles exhibiting the subyearling strategy compared to the fry, fingerling, and subyearling strategies, respectively.

Another distinguishing feature that salmon exhibit that influences their use of estuarine habitat is whether a juvenile is smolting compared to a juvenile that has yet to enter into the smolting process. Those animals having made the physiological transformation associated with the endocrine driven smolting process exhibit negative rheotactic behavior and typically use deeper water main channel habitats (associated with the stronger flow signals). Juvenile salmon that have not entered smoltification, but still are moving from natal rearing areas into the mainstem, estuary, and ocean habitats more frequently use side channel, shallow water habitats within the estuary. However, even for juvenile salmon that have smolted and have been shown to migrate through the estuarine habitat for short periods of time (days), a majority of them are found with prey items in their stomachs, suggesting they are extracting resources from estuarine habitats (Dawley 1989).

Although there are certain periods when large numbers of juveniles enter and utilize the estuary, current use patterns indicate that juvenile salmon use the estuary during the entire year (Figure 13). Obviously, existence of sufficient amount and subtypes of habitats allows all salmon species and steelhead in the Columbia River basin to express that appropriate spatial structure and diversity of life history strategies demanded by the environmental and biological conditions the juveniles encounter. This characteristic year long presence is consistent with the historical record. Burke (2001) reconstructed the presence of juveniles from research conducted by Willis Rich in the early 1900s (Figure 2). It is apparent that over the year, juvenile salmon representing different cohorts expressing varying life history strategies were using the Columbia River estuary.

Information on the size of salmon in the estuary also reveals more specifically how and where salmon use the estuary. During year long surveys that are actively ongoing to characterize salmon and steelhead entry and exit in the Columbia River estuary, the size structure of chinook salmon, as an example, show several features of how they use the estuary (Figure 14). During the early winter months, the size of juveniles entering and leaving the estuary are nearly identical, consistent with salmon exhibiting the fry life history strategy. However, as the size of chinook salmon entering the estuary increases in length as the year proceeds, the corresponding length of juvenile chinook salmon exiting the estuary during the same period are significantly larger, suggesting growth and thus rearing is occurring in the estuary.

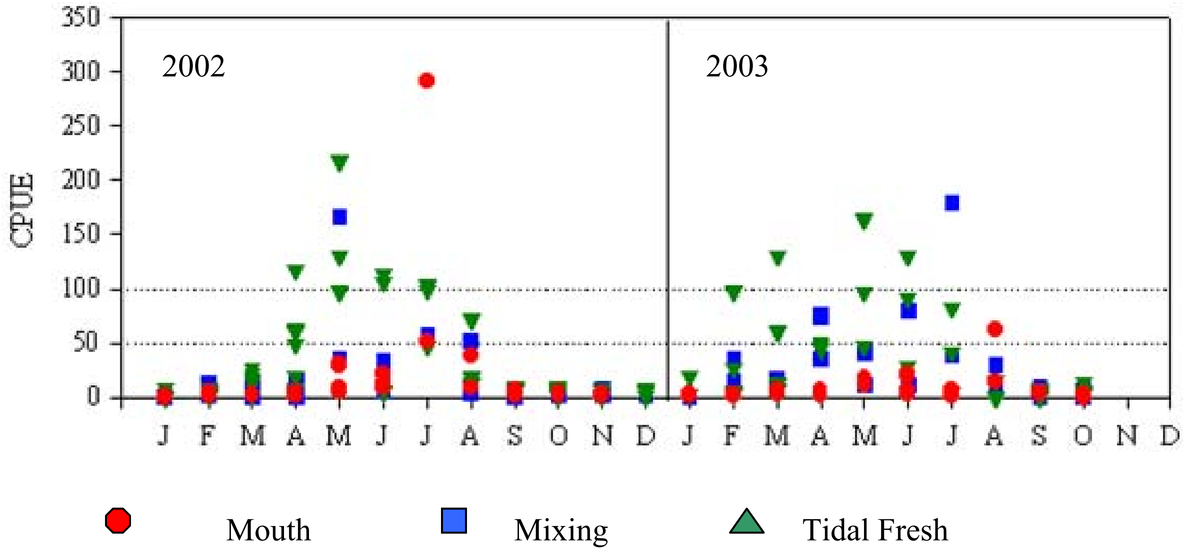


Figure 13. Catch per unit effort for juvenile chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (Curtis Roegner, NOAA Fisheries, pers. comm.)

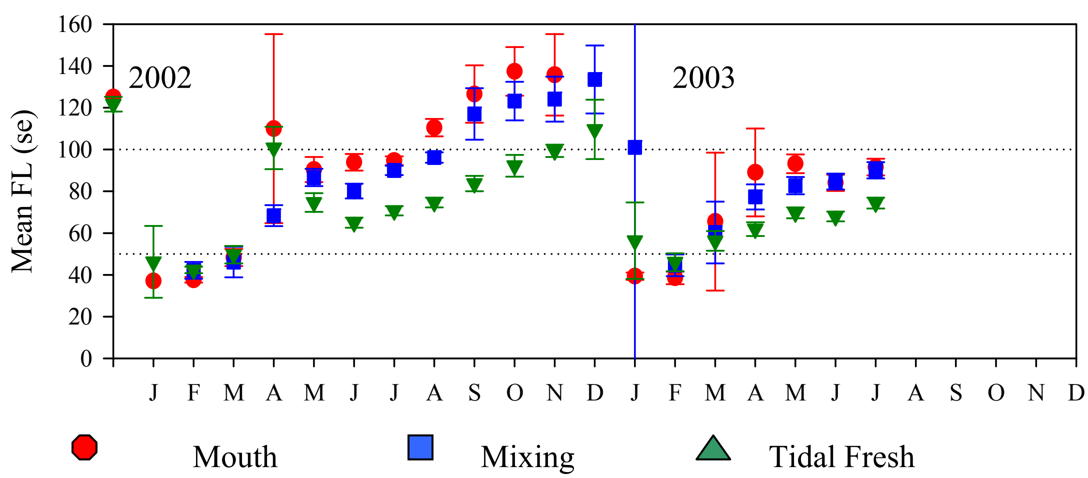


Figure 14. Mean fork length of juvenile chinook salmon for 2002 and 2003 at several sites in the mouth of the Columbia River estuary (circle), in the mixing zone (square), or in the tidal freshwater zone (Curtis Roegner, NOAA Fisheries, pers. comm.)

The different sizes of salmon observed in these ongoing studies and thus representing the different strategies employed, provide evidence of the differential use of specific types of habitat that salmon utilize. Smaller, unmarked chinook salmon, characterized as likely to be largely composed of naturally produced wild salmon are associated with side channel, peripheral tidal marsh and forested marsh habitats, whereas larger chinook salmon (characterized as hatchery releases) dominate the more deeper-oriented mainstem channel habitats
(Figure 15).

Although it has been argued by Bottom et al (2001) that salmon occupying shallow water habitats express the range of strategies characteristic of ocean-type salmon, it is now evident that salmon representing most of the endangered ESUs are using the peripheral habitats of the Columbia River estuary. Using genetic analysis employing recently developed microsatellite DNA which allows segregation of mixed populations of captured juveniles revealed that both ocean and stream type chinook from upper and lower basin sources were found in these marsh and forested wetland habitats (Figure 16, Paul Moran pers. comm.).

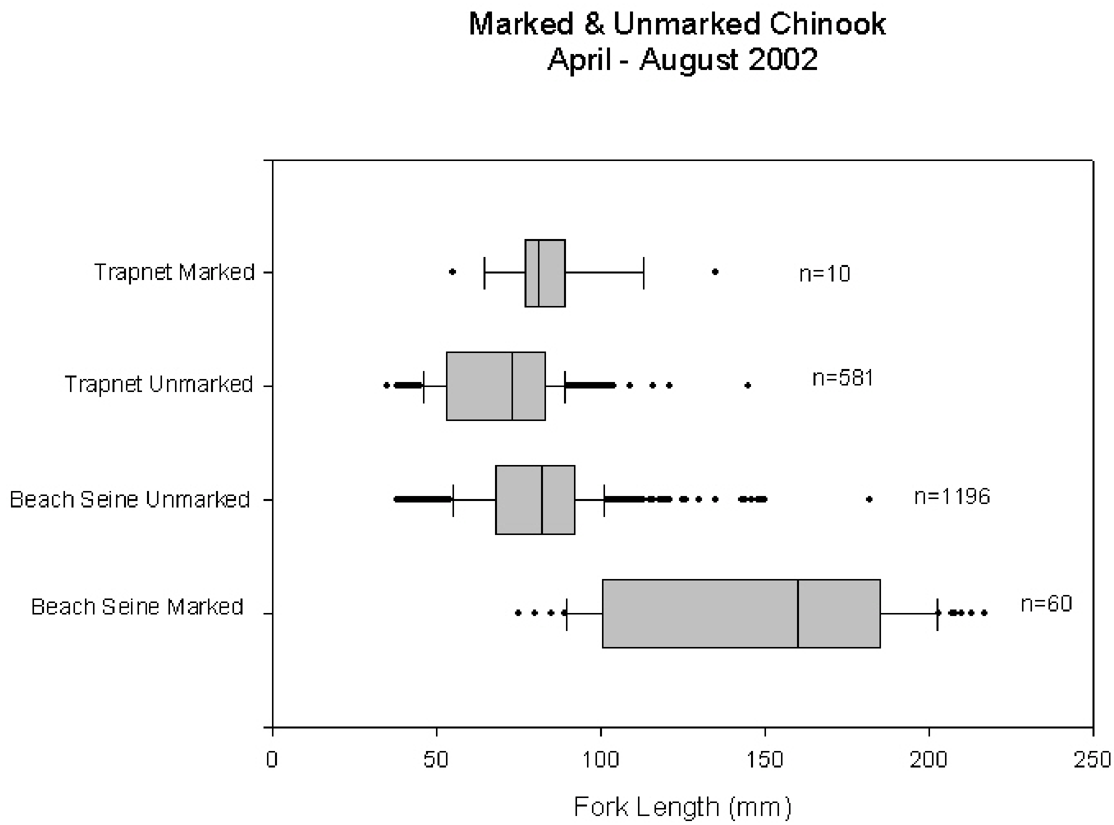


Figure 15. Box plots of size of juvenile salmon captured in peripheral habitats (trapnets) and near the main channels (beach seine) of the Columbia River estuary. Marked salmon (adipose clipped) represent hatchery fish whereas unmarked fish (adipose fin present) represent an unknown mixture of naturally produced and hatchery released juvenile salmon.

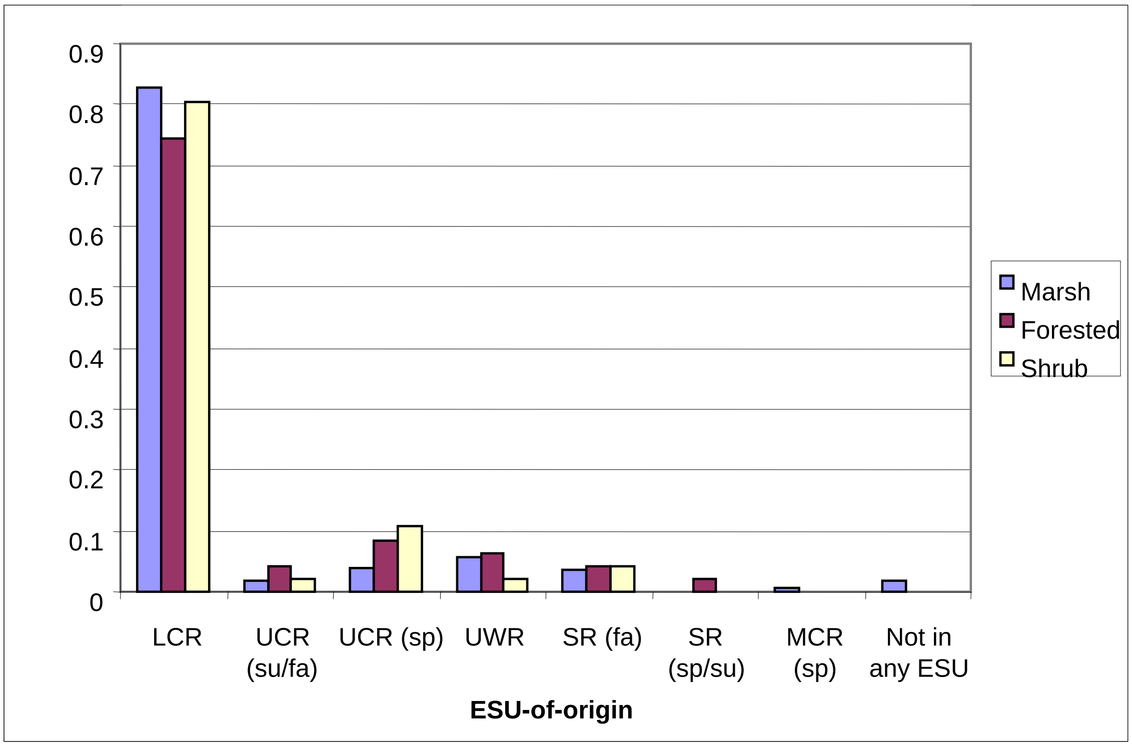


Figure 16. Proportion of Chinook salmon ESUs originating from various parts of the basin identified in samples taken from peripheral habitats of the Columbia River estuary during 2002 between April to August. LCR – Lower Columbia River, UCR – Upper Columbia River, UWR – Upper Willamette River, SR – Snake River, MCR – Middle Columbia River, su/fa – summer/fall run, sp – spring run, sp/su – spring/summer run. From Paul Moran, NOAA Fisheries, pers. comm..)

Although the predominant source of fish in marsh habitats were from the lower Columbia River Chinook ESU as predicted (Bottom et al. 2001), juvenile chinook from a variety of other source populations were evident. Confirmation that ESUs express a variety a strategies in estuarine habitats is gained from ongoing studies evaluating the role of the Columbia River plume as habitat for juvenile salmon. Spring chinook that express both yearling and subyearling strategies have been identified in the plume environment (Figure 17). Clearly, salmon expressing a variety of strategies from source populations originating throughout the basin use estuarine habitats throughout the year.

The characterization of how salmon employing varying strategies use the estuary is comparable to recent evidence developed regarding the role of the Skagit River estuary in Puget Sound, Washington, for salmon and salmon recovery. Casey Rice (NOAA Fisheries, pers. comm.) has developed empirical evidence that smaller, naturally developed wild salmon are present in the estuary for longer periods of time (Figure 18) and associated with the more peripheral oriented shallow water habitats than larger juvenile salmon either representing hatchery fish, earlier in the season, or wild fish growing larger and utilizing deeper water habitats later in the season (Figure 19). The value of this information stems from the similarity of both the Skagit and Columbia Rivers as large watershed systems with large historically floodplain dominated estuaries.

An important question to consider about estuarine habitat is whether habitat availability in the estuary can be a limiting factor to production and expression of a diversity of strategies. Although this information is forthcoming for the Columbia River estuary explicitly, studies in the Skagit River system have identified a density dependent limit to the number of juveniles in the estuary relative to the overall abundance of young salmon in the system (Figure 20). In the face of altered (i.e., reduced) habitat availability, it is clear that reducing the opportunity to access habitats at the appropriate time can be a limiting factor in production and recovery of depressed salmon and steelhead populations.

Finally, evidence that estuaries are likely to be important to recovery of endangered salmon and steelhead stocks is derived from salmon life cycle models and application of sensitivity analyses to identify critical life stages, and thus associated habitats, important to recovery. Kareiva et al. (2001) showed that improvement of survival of juvenile salmon during the estuarine and early ocean stage would significantly improve salmon population growth rates. Although they could not differentiate the contribution of the estuarine phase from the early ocean phase in their analysis, clearly the estuarine stage could be critical to recovery. Similarly, Greene et al. (in press) demonstrated that variability in conditions in the nearshore zone of Puget Sound, which serves as an extension of the Skagit Bay estuary, accounted for significant variability in

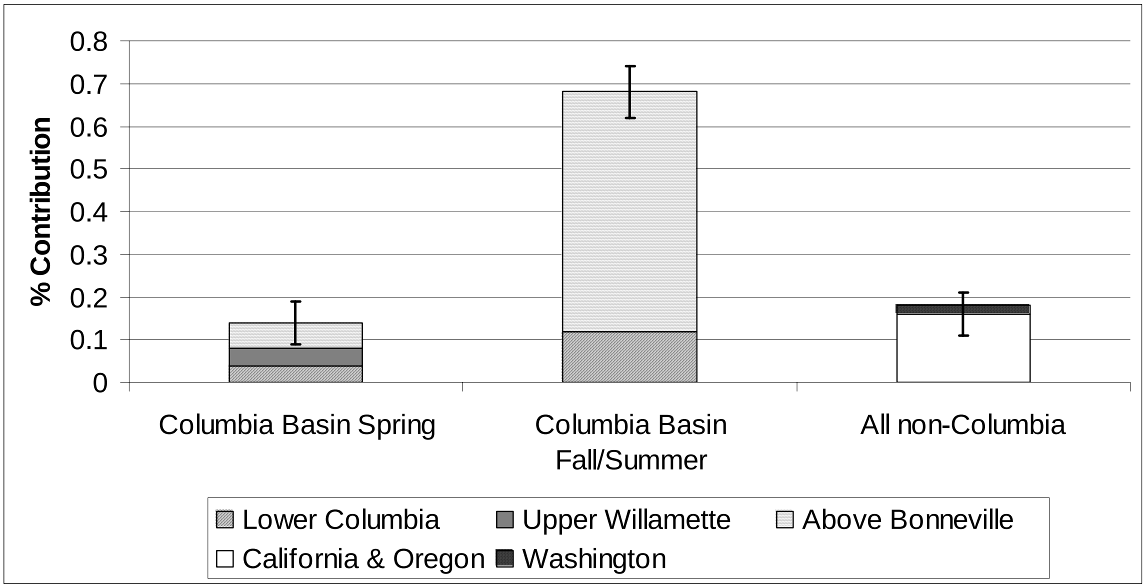


Figure 17. Stock composition of subyearling chinook salmon in Columbia River plume study area June 1998 – 2001.

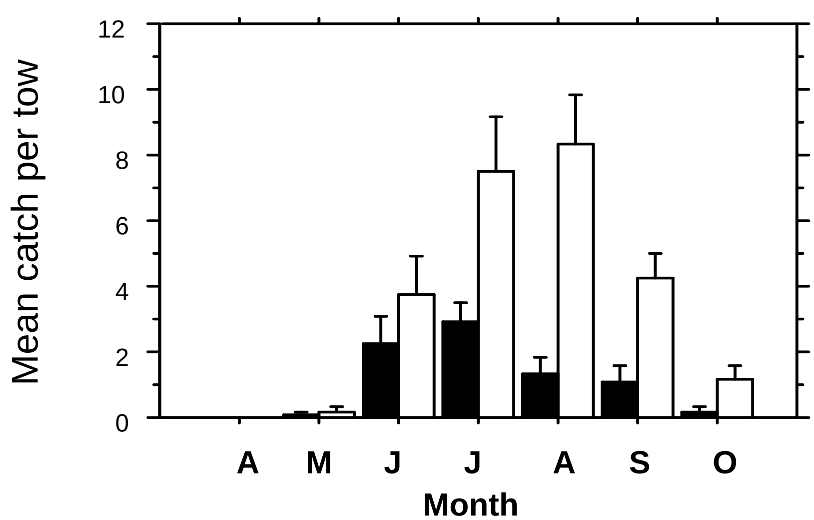


Figure 18. Temporal presence of naturally and hatchery produced juvenile chinook salmon in Skagit Bay, WA, an enclosed olgohaline region associated with the Skagit River watershed.

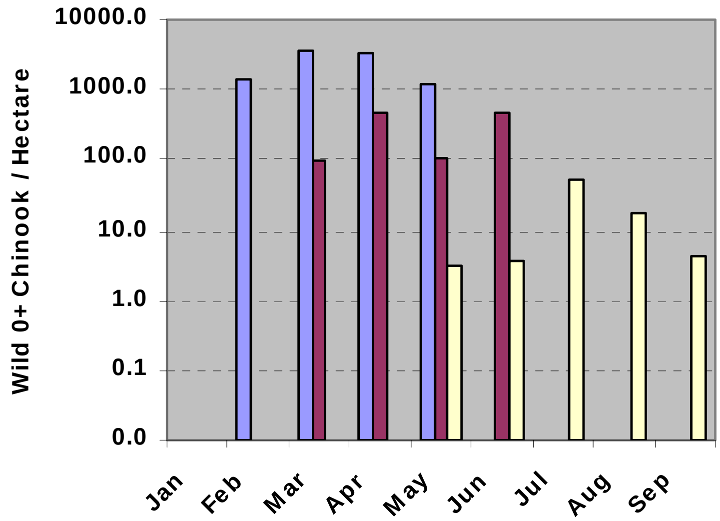


Figure 19. Movement of juvenile Chinook salmon in the Skagit River estuary system. Juveniles more prevalent in near shore habitat (blue), then move to deeper, shore oriented habitats later in the year, and more offshore habitats (yellow) towards the end of the year. The proportion of wild (unmarked salmon) ranged from 98%, 82%, and 73% in the peripheral, deeper, shore oriented, and channel habitats, respectively.

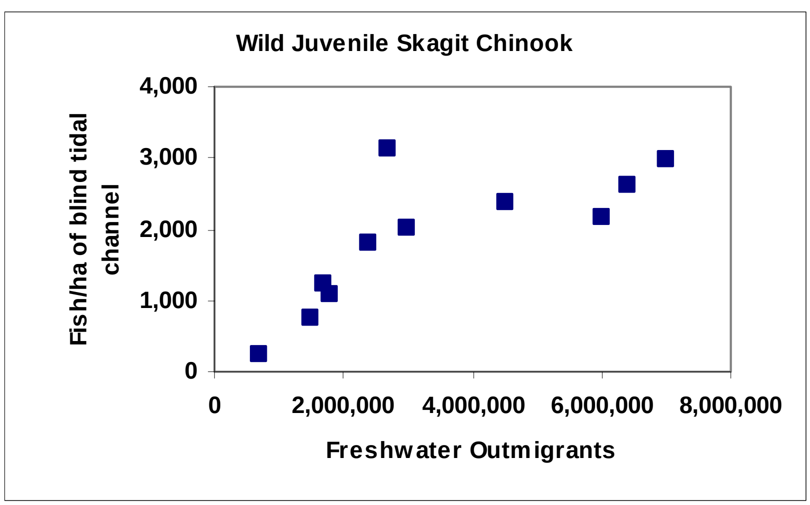
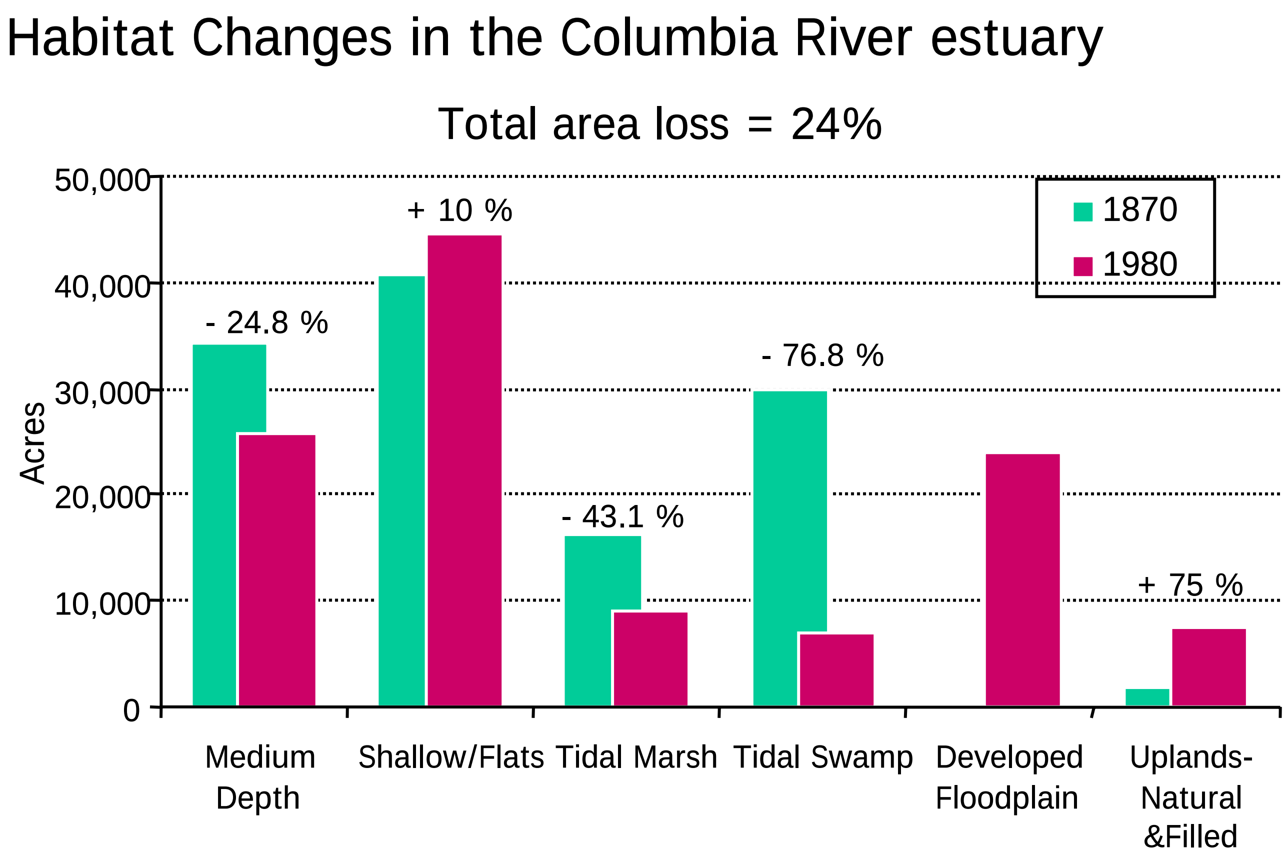


Figure 20. The relationship between freshwater wild chinook smolt population size and density of juvenile wild Skagit Chinook in Skagit River delta habitat, 1992-2002. The number of chinook per unit area within the delta levels-off as the total number of outmigrants increases, indicating density dependent use of the delta (E. Beamer. Skagit Bay Cooperative Res. Dept. Pers. comm).

adult returns for Skagit Bay chinook salmon. Incorporating density dependence, as indicated above, improved the resolution of the model output. Overall, the summary of information from a variety of perspectives documents that salmon use and benefit from the estuary and the habitats it contains. Identifying alterations that affect access or quality of the habitats relative to the historical record can be used to assess the potential of habitats within the estuary to represent a potential limiting factor of importance to recovery of endangered salmon populations.

Considering the importance of estuarine habitat to juvenile salmon, what evidence from the historical and more current record can be used to identify a change in the availability of habitat in general, or more specific, loss of salmon type habitat in the system. Thomas (1983) and Sherwood et al. (1990) have calculated losses of emergent marsh and forested wetland habitats in the Columbia River estuary. They characterize the change as substantial and likely a significant factor reducing the estuary's opportunity and capacity to support juvenile salmon. Approximately 121.6 km^2 of tidal marshes (77% decline) and swamps (62% decline) that existed prior to 1870 have been lost (Figure 21). Together with a 12% loss of deep-water habitat, these changes reduced the estuary's tidal prism from 12 to 20%. In addition, the historic surface area of the estuary has decreased by approximately 20% as a result of diking or filling of tidal marshes and swamps.

High elevation tidal marshes have been diked more than lower elevation marshes. New tidal marsh formation has resulted primarily from vegetative colonization of disposed dredge material. The location of tidal marsh habitat within the estuary has changed as a result of modified flow regime, modified tidal action, and/or shipping channel development and maintenance. Tidal swamp is the most impacted habitat type. Almost all the tidal swamp habitat present in 1870 was converted to diked floodplain/non-tidal habitat. Historically, few tidal swamps were present in the mainstem areas of the Columbia River estuary, in contrast to what has occurred in the peripheral bays. For example, there has been almost complete loss of all 1870 tidal swamp habitat from Youngs Bay and Baker Bay effectively eliminating brackish tidal water from the estuary. In the areas furthest upstream, losses of tidal swamps have been extensive, although a substantial amount of tidal swamp acreage is still present, particularly in the Cathlamet Bay region.



Data Source:Thomas, T.W.1983. Changes in Columbia River estuary habitat types over the past century. CREDDP

Figure 21. Change in acreage of various habitat types used by salmon in the Columbia River estuary from 1870 to 1980.

Johnson et al. (2003) in developing a restoration plan for the Columbia River estuary refined the losses within particular zones of the estuary using information from Thomas (1983), Graves et al. (1995), USACE (1996), and Garono et al. (2002). To facilitate this comparison, the Columbia River estuary and lower mainstem was delineated into eight distinct areas based on physical characteristics. A brief description of each estuary area, a summary of habitat changes, and a qualitative description of changes in select habitat characteristics were extracted from Johnson et al. (2003) and the subbasin review developed by the Lower Columbia Fish Recovery Board as follows:

Entrance–dominated by subtidal habitat; highest salinity in estuary; historically a high-energy area of natural fluvial land forms, a complex of channels, shallow water, and sand bars; supports the Columbia Plume; abrupt changes have resulted from dredging and jetty construction that limit the ocean-fed supply of sediment; impacts have manifested in increased deep water habitat (18.9%) and a loss of medium-depth (41.1%) and tidal flat (43.6%) habitat types.

Mixing Zone–characterized by a network of mid-channel shoals and flats; highest variation in salinity based on tide cycle and river flow; relatively little change in acreage of the five major habitat types.

Youngs Bay– haracterized by a broad flood plain and historically abundant in tidal marsh and swamp habitat; diking and flood control structures used to convert land to pasture resulted in 86.4% loss of tidal marsh and 95.7% loss of tidal swamp habitat. This subarea was responsible for the majority of lost tidal marsh habitat throughout the estuary.

Baker Bay–historically a high energy area from ocean currents and wave action; migration of mid-channel islands toward the interior of Baker Bay have sheltered the area; some tidal marsh habitat recently started to develop because of decreased wave action; potentially the most altered estuary area overall (-75.0% deep water, -71.3% medium depth, +74.9% tidal flats, -55.5% tidal marsh, and -100% tidal swamp habitat).

Grays Bay–pile dikes adjacent to the main Columbia River navigation channel have decreased circulation in the bay and caused flooding problems in the valley bottoms; accretion in the bay has led to development of tidal marsh habitat, increasing acreage 145.2% compared to historic conditions; dike construction for pasture conversion has isolated the main channel from its historic floodplain and decreased tidal swamp habitat 88.4% compared to historic acreage.

Cathlamet Bay–characterized by some of the most intact and productive tidal marsh and swamp habitat in estuary; large portion of area protected by the Lewis and Clark Refuge; other portions are heavily impacted by diking (Brownsmead area and Swenson Island) causing a 48.9% decline in tidal swamp; medium and deep water habitats have decreased (30.4% and 12.5%, respectively) as a result of dredge material disposal; fringe of dredge disposal areas has developed into tidal marsh habitat, resulting in 6.8% increase over historic acreage.

Upper Estuary–characterized by deep channels and steep shorelines on both sides of river; dominated more by tidal swamp habitat and less tidal marsh habitat; typically dominated by freshwater, except during low river flow or large flood tides; extensive diking and clearing has resulted in substantial loss of tidal marsh (64.3%) and tidal swamp (79.9%) habitat compared to historic acreage.

Tidal Freshwater–distinct in geology, vegetation, and climate; influenced by major tributaries; contains elongate islands that divide the river and form oxbow lakes, sloughs, and side channels; historically dominated by a combination of tidal plant communities, ash riparian forests, and marshy lowlands; historic data for entire area is limited to other areas so historic comparisons are not as robust; from rm 46-102, increased upland habitat in the middle reach and substantial loss of non-tidal water/wetland, tidal flats, and tidal marsh habitat types, with no comparison category for tidal swamp habitat; from rm 105-146, increased non-tidal water/wetland and upland habitat and substantial loss of tidal flats and tidal marsh habitat types, with no comparison category for tidal swamp habitat.

Thomas (1983), as reported in the Estuary Subbasin Plan (2003), also investigated five categories of non-estuarine habitat (i.e. developed floodplain, natural and filled uplands, non-tidal swamps, non-tidal marshes, and non-tidal water) to identify the fate of floodplain areas that were removed from the estuarine system. Developed floodplain habitat was defined as all diked floodplain converted to agriculture, residential, or other land use. Natural and filled uplands included areas where measurable acreages have been filled, primarily through disposal of dredge material. Non-tidal swamps were areas of the diked floodplain that were never cleared or were cleared and converted back to swamp. Non-tidal marshes included areas of the diked floodplain that support emergent wetland vegetation; these were typically abandoned pastures dominated by rush and sedge. Non-tidal water consisted of former tidal sloughs that were separated from the river by dikes and tidegates. The largest increase, by far, of non-estuarine habitat from 1870 to 1983 was that of developed floodplain habitat. Of the 36,970 total acres of lost estuarine habitat, 64.8% was converted to developed floodplain (Thomas 1983).

As above, absolute changes in habitat opportunity alone should not be used to directly infer changes in the capacity of the estuary to support salmon. For instance, despite considerable loss of emergent and forested wetlands in the estuary and associated declines in macrodetrital production, the total area of estuarine shallows and flats actually increased 7% between 1870 and 1980. This was independently substantiated by Sherwood et al. (1990), who estimated 68.4 x 10^6 m^3 net sediment gain within the estuary between 1868 and 1958. Areas of sediment increase include peripheral bays such as Cathlamet Bay and Grays Bay, which had shoaling rates of 0.61 cm yr^{-1} and 0.63 cm yr^{-1} and net volumetric increases of 76.2 x 10^6 m^3 and 19.1 x 10^6 m^3, respectively.

Loss of estuarine wetlands not only reduced the total amount of shallow rearing habitat available to young salmon but also altered the magnitude and character of habitat capacity. The resulting decline in wetland primary production eliminated approximately 15,800 mt carbon $year^{-1}$ (84%) of macrodetritus that historically supported estuarine food webs. This macrodetritus originated from the vascular and macrophytic plants and microscopic algae historically produced within the estuary's wetlands. However, these losses were accompanied by an increase of approximately 31,000 t carbon $year^{-1}$ of microdetritus from upriver sources, originating principally from increased phytoplankton production in the reservoirs behind the mainstem dams (Sherwood et al. 1990). Nevertheless, the shifts in the sources and types of detritus available may have altered estuarine food webs, including those leading to salmon. For example, the epibenthic-pelagic food web supported by microdetrital sources favors production of calanoid copepods and other pelagic organisms that typically are not consumed by juvenile salmon (Bottom and Jones 1990, Sherwood, et al. 1990).

As a result of loss of habitat, altering the spatial distribution of the food web may also be an important determinant of habitat capacity in the estuary. Whereas the macrodetrital food web was historically distributed throughout the lower river and estuary, the contemporary microdetrital food web is concentrated within the localized mid-estuary region of the estuarine turbidity maximum (ETM).

We have no objective means to quantify the ecological effects of the habitat shift from emergent and forested wetlands to shallows and flats. For example, no historic data are available for salmonid diet composition or stomach fullness within tidal wetlands to compare with other estuarine habitats. Although juvenile salmon may not directly benefit from the microdetrital food web, there is some evidence that they have higher stomach fullness in the mid estuary compared with other estuarine regions (Bottom and Jones 1990). One possible mechanism that has yet to be verified for the increased feeding rates is that enhanced detrital concentrations within the ETM may also stimulate secondary production in adjacent mid-estuary shallows and flats. However, we do know that prey production and salmon stomach fullness values are relatively high in protected flats compared with many estuarine habitats. Jones et al. (1990) found that the standing crop of benthic infauna in protected flats of the estuarine mixing region (approximately RM-7 to RM-21) was more than an order of magnitude higher (2.058 g m^{-2} AFDW (ash free dry weight) than benthic fauna standing crop in any of the other channel or unprotected flat habitats (0.098-0.136 g m^{-2} AFDW) within the same estuarine region.

One additional recent example, exemplifies the loss of habitat in the tidal freshwater region of the estuary, where we most lack the empirical evidence of change and contribution to expression of spatial structure and salmon life history diversity. Kukulka and Jay (2003) indicated that diking removed nearly 52% of the shallow water flood plain habitat in the tidally influenced freshwater zone of the estuary (Figure 22).

It is obvious from the analysis that removing dikes alone would restore considerable amounts of shallow water estuary habitats. Further, diking entirely removes habitat from the estuarine system, while other anthropogenic factors change estuary habitats from one type to another (Thomas 1983). The degree to which estuary habitat types have been affected by diking is directly proportional to elevation; thus, the highest elevation habitat type (i.e. tidal swamp) has been impacted by diking the most (Thomas 1983).

Mainstem estuary habitat in the Columbia River have for the most part been reduced to a single channel where floodplains have been reduced in size and off-channel habitat has been lost or disconnected from the main channel. Dikes prevent over-bank flow and affect the connectivity of the river and floodplain (Tetra Tech 1996); thus, the diked floodplain is higher than the historic floodplain and inundation of floodplain habitats only occurs during times of extremely high river discharge (Kukulka and Jay 2003). There is a critical level (i.e. the elevation of the diked floodplain) where water level must reach before substantial floodplain habitat are inundated (Kukulka and Jay 2003). Above this critical water level, large amounts of shallow water floodplain habitats become available with small increases in water level up to an optimum threshold (Kukulka and Jay 2003).

Under a modern bathymetry and flow regime scenario, the critical river discharge level in which significant shallow water habitats become available through floodplain inundation is relatively high and the frequency of occurrence of this river discharge is rare; thus, floodplain inundation is uncommon and availability of shallow water habitats is limited (Kukulka and Jay 2003). As is the case in the estuary (Bottom et al. 2001), loss of these vital mainstem floodplain habitats has likely reduced the productive capacity of the lower Columbia River for juvenile salmonids, particularly those juveniles employing life history strategies associated with these peripheral, shallow water habitats.

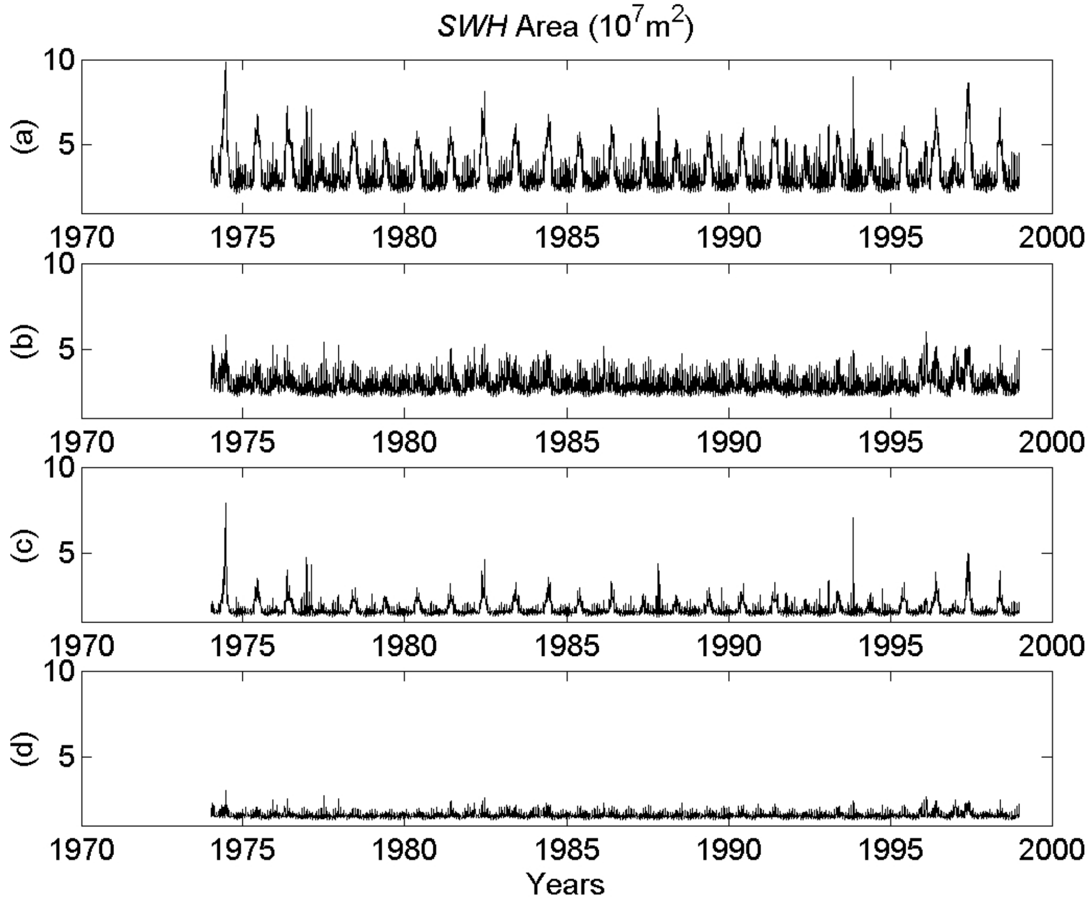


Figure 22. Daily Shallow-Water habitat (SWH) Area from 1974 to 1998 for virgin (a) and observed (b) river flows without dikes, and for virgin (c) and observed (d) flows with dikes, from Kukulka and Jay (2003).

Substantial evidence indicates that the productive capacity of the Columbia River estuary has declined over the last century. The results generally illustrate that the habitat opportunity and capacity of the Columbia River estuary may have declined through reductions in the estuarine tidal prism, surface area, and the amount of peripheral wetland habitat. Moreover, changes in the detrital sources that support estuarine food webs have affected competitive and likely subsequent predatory interactions in the estuary with uncertain, but potentially significant, consequences for salmon survival.

Many of the changes in the biological production processes of the estuary described above can be attributed directly to physical causes. For example, the apparent shift from macrodetrital to microdetrital food chains in the estuary stems from the diking and filling of intertidal wetlands and the creation of deep reservoirs behind mainstem dams. While changes in the quality and quantity of prey resources could well be a proximal factor affecting the productive capacity of the estuary, the ultimate cause is the physical removal of those habitats that supported both macrodetrital production and the diversity of estuarine life histories among salmon.

This is not to say that all biological effects are physically driven or somehow irrelevant to salmon survival and abundance. Nonetheless, we find that many of the significant biological changes we now observe in the Columbia River estuary are best explained by physical modifications that have altered the habitat landscape to the benefit and detriment of different species and assemblages. These findings have important implications for developing restoration strategies that address the ultimate causes rather than the proximal symptoms of salmon decline.

In summary, the location and types of habitats present in the Columbia River Estuary have been substantially changed from historic conditions. Although the entire estuary has not yet been surveyed, the main changes that have been quantified in the estuary have been a loss of emergent marsh, tidal swamp, and forested wetlands. Shallow water dependent life history strategies (fry and fingerlings) have been most affected by the loss of these vegetated habitat types. Alterations in attributes of flow and diking have caused these changes. Diking is a significant change primarily because it completely isolates habitat from the river and eliminates it from use by juvenile salmon. Further, it has altered estuarine food webs from macrodetrital to microdetrital based. Clearly, restoration of shallow water vegetated habitat by removing dikes is a tactic that can benefit those populations that have large numbers of shallow water dependent members.

Toxics

In addition to eliminating and physically altering salmon habitat in the estuary, the quality of habitats has been degraded through the release of toxic contaminants. With the exception of some metals and natural products, concentrations of toxic contaminants in the Columbia Estuary were historically low. However, beginning in the early 1800s, activities such as agriculture, logging, mining, industrial discharges, and stormwater runoff began to degrade water quality in the Columbia River estuary. Currently, the Lower Columbia from Bonneville to the estuary mouth is the most urbanized section of the river, encompassing the major urban centers of Portland and Vancouver and host of minor cities such as Longview and Astoria. The estuary receives contaminants from over 100 point sources (Fuhrer et al. 1996), as well as numerous non-point sources such as surface and stormwater runoff from urban and agricultural areas.

The largest sources of effluent are the Portland and Vancouver sewage treatment plants and associated combined sewer overflows in the Willamette River and the Columbia River Slough (LCREP 1999). Contaminants may also be transported to the estuary from areas of known sediment contamination above the Bonneville Dam such as the Yakima River (Rinella et al. 2000; Fuhrer et al. 1996), Lake Roosevelt (Bortleson et al. 1994) and other tributaries (Fuhrer 1989; Roy F. Weston, Inc.1998). Spills or other accidental releases of toxic substances at Bonneville Dam itself may also contribute to contamination in the estuary (**REFERENCES**), although inputs are probably relatively small compared to those from the large urban centers.

A number of potentially toxic water-soluble contaminants have been detected in the Lower Columbia River Basin. The USGS NASQAN program has reported a wide range of current-use pesticides in the water column at its Lower Columbia River sampling sites at Warrendale at RM 141 near the Bonneville Dam; the confluence of the Willamette and Columbia rivers near Portland at RM 101.5; and the Beaver Army Terminal at RM 53.8 (Fuhrer et al. 1996; Hooper et al. 1997). These water soluble contaminants include simazine, atrazine, chlorpyrifos, metolachlor, diazinon, and carbaryl. Water concentrations and frequency of detection were highest at the Willamette/Columbia confluence, with detections in 80-100% of samples at concentrations up to 300 ng/L; these compounds were also frequently reported at the Beaver Army terminal.

Various trace metals have also been monitored as part of this program, revealing high concentrations of iron and manganese, especially near the Willamette/Columbia confluence, and high levels of arsenic in the Lower Columbia (Fuhrer et al. 1996). These compounds come partly from natural sources, but also historic anthropogenic activities, such as the use of lead arsenate as an insecticide for apples. Concentrations of other trace metals were similar to background concentrations in other North American streams (Fuhrer et al. 1996).

Contaminants that have been documented in estuary sediments include trace metals (cadmium, copper, and zinc), dioxins, furans, chlorinated pesticides and other chlorinated compounds (e.g., dieldrin, lindane, chlordane, PCBs, and DDT and its metabolites), and polycylic aromatic hydrocarbons (PAHs) and other semi-volatile compounds (Fuhrer and Rinella 1983; Fuhrer 1986; Harrison et al. 1995; Tetra Tech Inc 1996; US Army Corps of Engineers 1998; Roy F. Weston, Inc 1998). Many of these compounds, particularly lipophilic compounds such as PAHs and organochlorine compounds (OCs), are rarely detected as dissolved material in the water, but rather tend to bind to organic carbon or particulate materials; thus, they commonly occur in association with fine-grained materials in the streambed or in suspension (Horowitz 1991; Tetratech Inc. 1993).

The fine-grained sediments to which these toxicants adsorb will most likely be deposited in areas with slower water velocities, including backwater areas in side channels and along the river's margins, so elevated concentrations of toxic contaminants are more likely to occur in these areas (Tetra-Tech, Inc. 1994). In the main navigation channel of the Columbia River, current velocities are generally high, so there is little deposition of fine-grained sediments. Coarse, sandy sediment with relatively low contaminant concentrations typically make up 99% of the bulk bed material in the navigation channel (USACE 1998, 1999; McCabe et al. 1997).

Some contaminants, including PAHs, have been detected at levels that exceed State or Federal sediment quality guidelines or are considered harmful to humans and aquatic life (Tetra Tech Inc. 1996). These contaminants have been detected in sediments from the lower Willamette River in the Portland area (concentrations up to 900 mg/kg wet wt (ODEQ 1994a; Harrison et al. 1995; Roy F. Weston Inc. 1998) and in some sediments from other near urban and industrial areas in the estuary (Tetra Tech, Inc. 1996). Recent sediment data, collected in 2000 by EPA as part of the EMAP program, identified a few hot spots for PCBs and DDTs within the estuary, including sites near Longview (total PCBs 860 ng/g dry wt); West Sand Island (total PCBs 965 ng/g dry wt), the Astoria Bridge (DDTs 597 ng/g dry wt), and Vancouver (DDTs 128 ng/g wet wt).

It is noteworthy that PCB and DDT concentrations in the majority of sediments tested in these studies were much lower (below 50 and 5 ng/g dry wt, respectively). To put these values in perspective, the sediment screening guidelines for the protection of marine life are typically in the 20-200 ng/g dry wt range for PCBs (MacDonald 1994;

USACE 1998; **WAC ???**; Meador et al. 2002; CCME 2002), and in the 3-7 ng/g dry wt range for DDTs (MacDonald 1994; USACE 1998; WAC 173-204; Meador et al. 2002; CCME 2002), depending on the organic carbon content of the sediment.

Suspended material may also be an important source of contaminants in the Lower Columbia (LCREP 1999). This material is predominantly fine-grained, and contains many of the toxic compounds that have been detected in streambed sediments. McCarthy and Gale (2001) analyzed water samples collected with semi-permeable membrane devices from nine main-stem and six tributary sites throughout the Columbia River Basin (Washington and Oregon) and found dioxins, dibenzofurans, PCBs, organochlorine pesticides, and PAHs throughout the basin, with highest concentrations of many compounds in the Portland-Vancouver area. Metals including arsenic, lead, chromium, copper, iron, manganese, mercury, and zinc have also been detected in suspended sediments in the estuary (Fuhrer et al. 1996).

Mobilization and transport of suspended sediments during extreme stream flow events can make their adsorbed contaminants available to salmon and other aquatic organisms. For example, during the flood of February 1996, several legacy organic pesticides, including dieldrin and DDE, that are typically associated with the sediment phase were mobilized in the Lower Columbia and Willamette Rivers and detected in the water column at some sites for the first time (Kelly 1997). During this event, the estimated whole water concentration of p,p'-DDE exceeded the chronic ambient water-quality criterion for the protection of aquatic organisms by at least five-fold. Suspended particulates and associated contaminants may also occur in areas of high turbidity, such as the estuarine turbidity maximum, which may be an important feeding area for salmon (Bottom and Jones 1990). However, currently, the relative contributions of contaminants in water column vs. those in bed sediment to body burdens in resident biota are poorly understood (LCREP 1999; SEI 2001).

Exposure to contaminants, and hence the potential effects of these compounds, in the estuary likely varies with life history type or ESU. Stream type ESUs (e.g., Snake River sockeye, Upper Willamette steelhead, and Snake River spring chinook) are less likely to accumulate high body burdens of bioaccumulative, sediment-associated contaminants such as PCBs and DDTs because most members of these populations migrate rapidly through the estuary as yearlings or older. However, they may be affected by short-term exposure to waterborne contaminants such as OPs and dissolved metals.

Ocean type fish (e.g., Lower Columbia River chum and Upper Willamette River Chinook), which enter the estuary as fry, fingerlings, or subyearlings, and may rear for an extended period in the estuary, are also at risk for exposure to current use pesticides and dissolved metals. At the same time, they are more likely than stream type fish to be affected by bioaccumulative toxicants (DDTs, PCBs) that they may absorb through their diet during estuarine residence. Ocean-type populations may also be more at risk because of their greater use of shallow-water habitats with slower water velocities, including backwater areas in side channels and along the river's margin, where fine-grained sediments to which toxics adsorb are most likely to accumulate. This may be particularly true for naturally produced wild salmon that, as noted earlier in this document, are more commonly found in side-channel, peripheral tidal marsh and forested marsh habitats than the larger, hatchery size salmon that mostly use deeper mainstem habitat.

Many of the contaminants found in streambed and suspended sediments in the estuary are accumulated by resident biota. A number of studies have identified trace metals, dioxins and furans, chlorinated compounds such as PCBs and DDTs, and PAH metabolites in non-salmonid fish from the Lower Columbia (Tetra Tech Inc 1993d, 1996; Brown et al. 2000; Foster et al. 2001a,b), in some cases at concentrations have exceeding health guidelines (LCREP 1999). Although data on contaminant concentrations in listed salmon from the Lower Columbia are more limited, they indicate that chemical contaminants are present in prey and tissues of juvenile salmon from the Columbia Estuary (Figures 23-26).

In studies conducted by the Northwest Fisheries Science Center in collaboration with the Army Corps of Engineers (Johnson et al. 2004), contaminant concentrations were measured in juvenile fall Chinook salmon from several sites in the estuary (near the confluence of the Columbia and Willamette Rivers, near Longview, White Island, West Sand Island, between East and West Sand Island, Chinook Point, East Trestle Bay, West Trestle Bay, Lower Desdemona Sands, and Middle Desdemona Sands; Figures 23-26). Fish from the Wilamette/Columbia River confluence, Longview, and West Sand Island were collected in shallow water habitats by beach seine, while fish from the other sites within the Lower Estuary were collected in deeper water by purse seine.

The primary contaminants found in whole body samples of both purse seine and beach seine fish from all sites were PCBs and DDTs. Chlordanes, lindane, hexachlorobenzene, dieldrin, and mirex were also detected in fish from the confluence and Longview/Kalama. Average concentrations of PCBs at estuarine sampling sites ranged from 23 to 90 ng/g wet wt), while average DDT concentrations ranged from 32 to 115 ng/g wet wt). In individual fish, DDT levels as high as 270 ng/g wet wt and PCB levels as high as 340 ng/g wet wt were measured. These concentrations were among the highest levels measured by the NWFSC at estuarine sites in Washington and Oregon (M. Arkoosh, NWFSC, unpublished data; Collier et al. 1998a, Stehr et al. 2000; Stein et al. 1995).

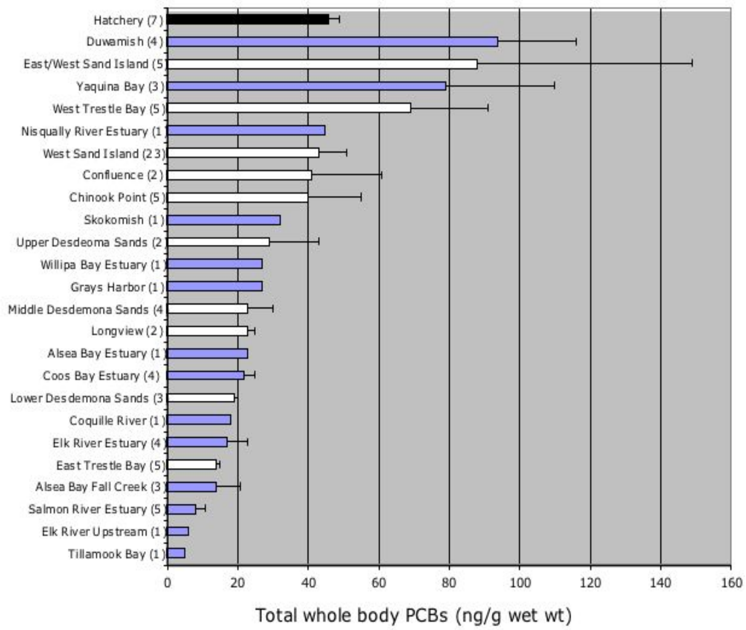


Figure 23. Mean concentrations (± SE) of total PCBs (ng/g wet wt) in whole bodies of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia Estuary are indicated in white.

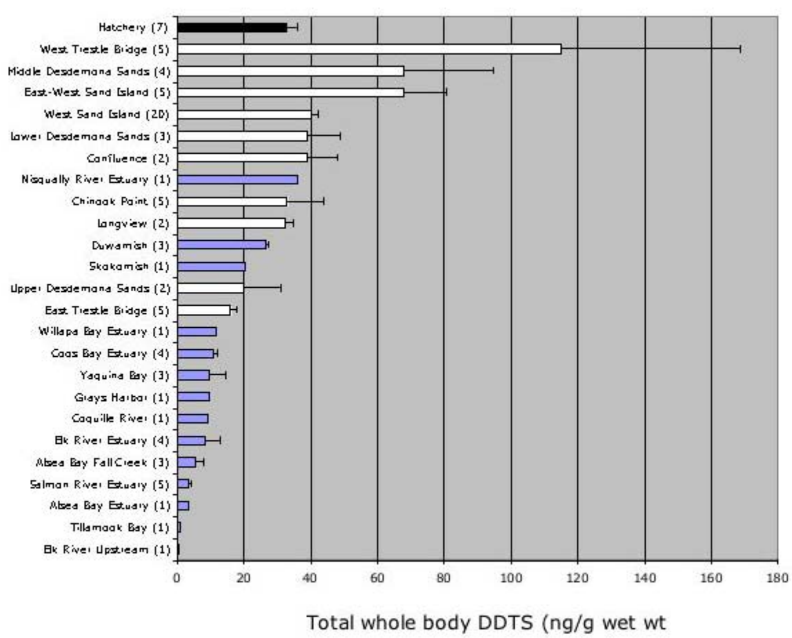
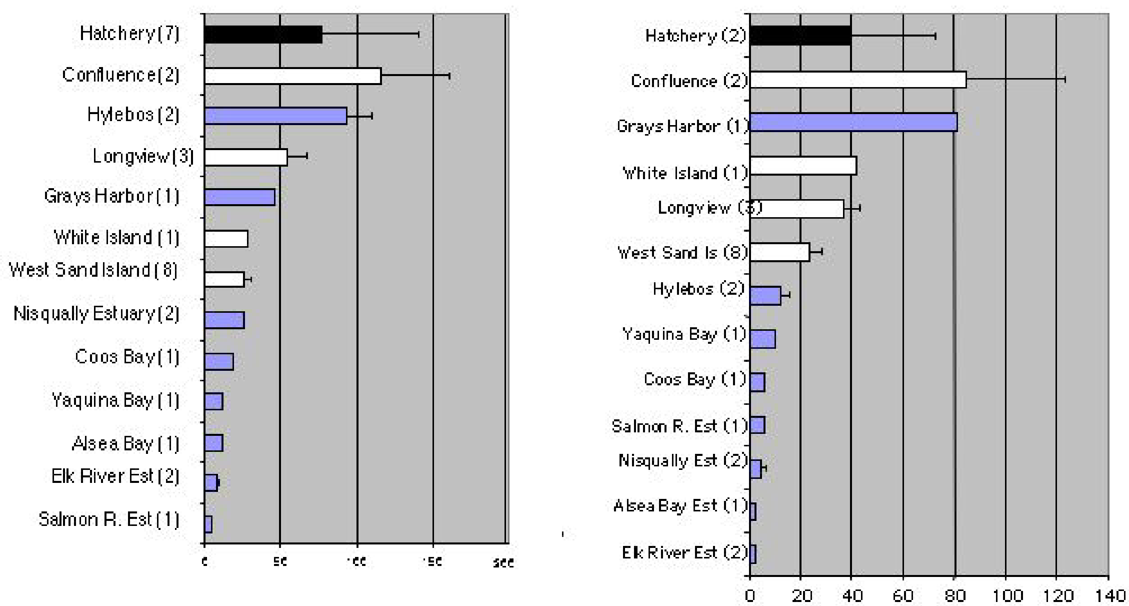


Figure 24. Mean concentrations (± SE) of DDTs (ng/g wet wt) in whole bodies of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia Estuary are indicated in white.



PCBs in stomach contents
(ng/g wet wt)

DDTs in stomach contents
(ng/g wet wt)

Figure 25. Mean concentrations (± SE) of PCBs and DDTs (ng/g wet wt) in stomach contents of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia are indicated in white.

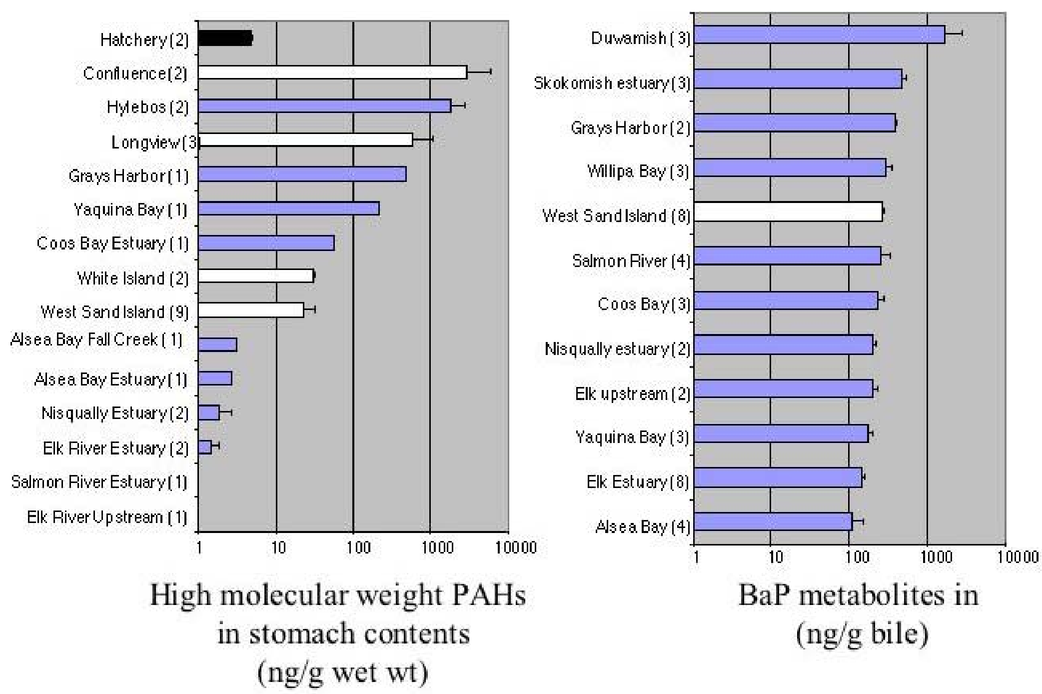


Figure 26. Mean concentrations (± SE) of high molecular weight polycyclic aromatic hydrocarbons (PAHS) (ng/g wet wt) in stomach contents and metabolites of PAHs in bile of juvenile fall Chinook salmon sampled from Pacific Northwest estuaries. Sites from the Lower Columbia are indicated in white.

Measurable concentrations of DDTs and PCBs, were also detected in stomach contents of juvenile fall Chinook salmon from West Sand Island, White Island, Longview, and the Columbia-Willamette Confluence (at other sites, contaminants in stomach contents were not measured), indicating that fish were absorbing some compounds from prey during estuarine residence. Contaminant concentrations in fish from the Willamette/Columbia confluence are shown in Figure 23. Concentrations of PCBs in stomach contents of fish from this area were comparable to those from in fish from the heavily industrialized Hylebos Waterway in Puget Sound (Stehr et al. 2000), while DDT concentrations were higher than at any other sampled sites in the Pacific Northwest. Several additional chlorinated pesticides, including lindane, hexachlorobenzene, dieldrin, and certain DDT isomers (o,p-DDD, o,p-DDT, p,p-DDT) were detected only in stomach contents of salmon from the confluence or Longview.

Although the fish used for these analyses were almost entirely unmarked, genetic analyses showed that a high proportion of beach seine collected fish from West Sand Island, Longview, and the confluence were most likely of hatchery origin (40-70%; Johnson et al. 2004). Consequently, DDTs and PCBs in hatchery feed may have contributed to contaminant body burdens in these fish, since these substances, especially PCBs, have been detected in hatchery feed and in juvenile Chinook salmon collected from Pacific Northwest hatcheries (M. Arkoosh, and G. Ylitalo, NWFSC, unpublished data; Johnson et al. 2004; Figures 23-26). However, no data are available on contaminant concentrations in feed from the specific hatcheries where these fish originated (predominantly hatcheries on the Elokoman, Cowlitz and Sandy rivers), so this cannot be confirmed. In spite of the potential for hatchery contribution, it is clear from the elevated levels of DDTs and PCBs in stomach contents of fish from sites within the estuary that fish are also being exposed to these contaminants through their natural prey. Additionally, DDT/PCB ratios were several times higher in bodies of salmon from Columbia River sites (average 1-4) than in hatchery fish (average 0.7), suggesting uptake of DDTs from the environment.

Less information is available on exposure to PAHs in juvenile salmon from the Columbia River estuary. Data collected by NOAA Fisheries between 1998 and 2002 showed that concentrations of PAHs in stomach contents and PAH metabolites in bile were low to moderate in juvenile fall Chinook from West Sand Island in comparison to levels found in fish from other estuaries along the Washington and Oregon Coast (Figure 26), but no data are available on metabolites of PAHs in bile of fish from other sites in the estuary.

PAHs were measured in stomach contents of juvenile Chinook salmon from several Lower Columbia sites (the Willamette/Columbia confluence, West Sand Island, White Island, and Longview). While levels in fish from West Sand Island and White

Island were moderate, concentrations in fish from Longview and the confluence were higher than or comparable to concentrations in juvenile salmon from the Hylebos Waterway in Puget Sound (Figure 26).

Data on contaminant concentrations in salmon prey and other benthic invertebrates are limited. Little has been published for contaminants in aquatic macroinvertebrates, although some work has been done on clams, crayfish, and *Corophium*, a benthic amphipod that is important in the diets of several fish species, including salmonids (McCabe et al. 1986, 1993; Muir and Emmett 1988; Willis CF 1997; LCREP 1999). A small-scale reconnaissance study conducted by NOAA Fisheries Newport laboratory in the mid-1990s measured PAH concentrations in *Corophium salmonis*[1] from several sites in the Lower Columbia, including Longview Bridge and the Multnomah Channel in the Columbia River, and sites at the Willamette River mouth, North Portland Harbor, and Hayden Island in the Willamette River.

At the three sites where sediment PAH concentrations were highest (Hayden Island, Longview Bridge, and Multnomah Channel), no amphipods were present in sediments, suggesting *C. salmonis* has specific habitat requirements, including sediment particle size preferences, and, possibly, concentrations of contaminants, that may prevent them from occupying these sites. PAH concentrations in tissues of *Corophium* tissues were up to over 50 ng/g wet wt BaP equivalents or about 150 ng/g wet wt HAHs. Average concentrations of HAHs in stomach contents from juvenile salmon for the Lower Columbia ranged from ~3000 ng/g wet wt and ~600 ng/g wet wt the Confluence and Longview, respectively, to 20-30 ng/g wet wt at West Sand Island and White Island (Johnson et al. 2004).

In general, these studies show that PCB and PAH concentrations in salmon or their prey from the Columbia River estuary are comparable to those reported in juvenile salmon from other moderately to heavily urbanized sites, while DDT levels are high relative to levels in other Pacific Northwest estuaries. Although concentrations of contaminants were higher in stomach contents of salmon juveniles collected from near Columbia/Willamette confluence, body burdens of bioaccumulative OCs were similar throughout estuary. Sources and pathways of exposure are unclear, and could include contaminated bed sediments, contaminated prey, and contaminants in suspended material, as well as hatchery feed for those fish that are of hatchery origin.

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[1] Data are from preliminary studies conducted by Dr. Bruce McCain, Hatfield Marine Science Center, Newport, Oregon. Concentrations of PAHs in amphipods were measured using a semi-quantitative method that employs high performance liquid chromatography (HPLC) and a photo-diode array (PDA) detector (Krahn et al. 1993).

Studies suggest that, at least for some contaminants, exposure levels in juvenile salmon from the Columbia River estuary are approaching concentrations that could affect their health and survival. For PCBs, Meador et al. (2002) estimated a critical body residue of 2400 ng/g lipid for protection against 95% of effects ranging from enzyme induction to mortality in a fish with 2% lipid, based on a range of sublethal effects observed in salmonids in peer-reviewed studies conducted by NMFS and other researchers. Mean PCB body burdens in juvenile salmon analyzed by the NWFSC (Johnson et al. 2004) were at or above these thresholds at several sites in the Lower Columbia (Figure 23). Of individual fish analyzed from sites within the estuary, ~35% were above the effects threshold.

The likely impact of DDTs, another major contaminant of concern in the Lower Columbia, on salmon is less clear. Most reported effects of DDTs on salmonids are associated with whole body tissue concentrations above those typically found in salmon obtained from the Columbia estuary (≥500 ng/g wet wt) (Allison et al. 1962; Burdick et al. 1964; Johnson and Pecor 1969; Buhler et al. 1969; Peterson 1973, Poels et al. 1980; Hose et al. 1989). More recent studies suggest that certain DDT isomers (e.g., o,p-DDT) may have endocrine-disrupting or immunotoxic effects (Donohoe and Curtis 1996; Celius and Walther 1998; Khan and Thomas 1998; Arukwe et al. 1998, 2000; Christian et al. 2000; Zaroogian et al. 2001, Milston et al. 2003; Papoulis et al. 2003, **Metcalfe et al. 200X**).

However, effects were typically observed at body burdens or dietary exposure concentrations in the 10-20 ng/g wet range or above, while concentrations of o,p-DDTs in the bodies and stomach contents of juvenile Chinook salmon from the Lower Columbia were near or below detection limits (< 2 ng/g wet wt; Johnson et al. in prep). This suggests that o,p-DDT, DDE, and DDD levels are below concentrations likely to be associated with estrogenic activity and related effects. These compounds could work in conjunction with other estrogenic contaminants (e.g., plasticizers, pharmaceuticals, and surfactants) to alter reproductive processes or other physiological functions, if these are also present in the estuary.

DDTs also have the potential to affect salmonid prey, as invertebrate species are generally quite susceptible to their impacts. Results of laboratory and field investigations, as well as equilibrium partitioning calculations, suggest that thresholds for chronic effects occur at total DDT concentrations in sediments of approximately 10 ng/g dry wt (Pavlou et al. 1987; Long et al. 1995). These DDT concentrations are not uncommon in esutary sediments (EMAP 2000). Moreover, studies in the Columbia Estuary have shown that DDTs represent a hazard to fish-eating predators through bioaccumulation and bioconcentration (Anthony et al. 1993; U.S. Fish and Wildlife Service 1999; Thomas and Anthony 1999, 2003; Henny et al. 2003). Recently, Nendza et al. (1997) estimated a no-observable effect concentration of 22-50 ng/g wet wt for food-chain related impacts for DDTs, based on studies with a number of fish from marine estuaries. Many juvenile salmon sampled from the Lower Columbia have DDT body burdens at or above this level.

The potential for contaminant-related injury to juvenile salmonids in the estuary is also supported by field studies at Puget Sound sites contaminated with PAHs, PCBs, and other OCs similar to those present in the Lower Columbia. In these studies, juvenile salmon from two urbanized waterways, the Hylebos and the Duwamish, showed demonstrable effects, including immunosuppression, reduced disease resistance, and reduced growth rates, due to contaminant exposure during their estuarine residence (Arkoosh et al. 1991, 1994, 1998; Varanasi et al. 1993; Casillas et al. 1995a,b, 1998a). For example, juvenile chinook salmon from the Duwamish Waterway were not able to produce the normally enhanced secondary immune response observed in non-exposed control fish from the Nisqually estuary and fish from the natal hatcheries (Arkoosh et al. 1991; Figure 27).

Salmon exposed in the laboratory to PCBs and PAHs were also immunosuppressed (Arkoosh et al. 1994). Additionally, in disease challenge studies with *Vibrio anguillarum*, a marine bacterial pathogen that infects juvenile chinook salmon from estuaries along the Washington and Oregon coast (Arkoosh et al. 2004), juvenile chinook salmon from the Duwamish Waterway were more susceptible to disease and exhibited higher cumulative mortality than fish from natal hatcheries on a non-contaminated estuary (Arkoosh et. al. 1998; Figure 28). Similar effects were found in juvenile chinook salmon injected in the laboratory with extracts of sediments from the Hylebos Waterway in Commencement Bay (Arkoosh et al. 2000).

In related studies designed to assess the effects of contaminants on growth (Casillas et al. 1995a,b), juvenile fall chinook salmon collected from the Duwamish Waterway and held in the laboratory for up to 90 days did not grow as well as similarly-treated fish from the natal hatchery on the Green River. In contrast, juvenile salmon from the non-urban estuaries showed no difference in growth compared to fish from the natal hatcheries. Furthermore, concentrations of insulin-like growth factor, a plasma hormone involved in the regulation of growth, were lower in fish from the urban estuary than in fish from the corresponding hatchery or the non-urban estuaries and hatcheries. In a separate laboratory study (Casillas et al. 1998a), growth was reduced in juvenile chinook salmon exposed to PCBs and to extracts of sediments from the Hylebos Waterway, another urban estuary. Because growth of salmon during the first year of life appears to be critical to recruitment success (Holtby et al. 1990; Pearcy 1992; Unwin 1997), contaminant-related alterations in juvenile growth rates could increase the risk of salmon mortality.

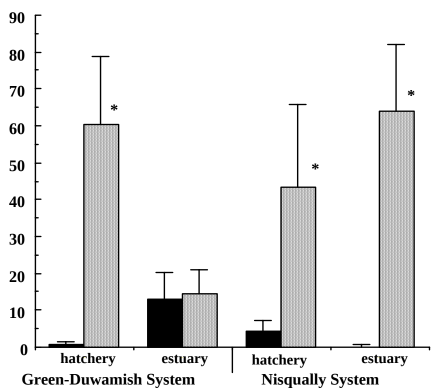

Figure 27. The leukocyte primary (■) and secondary (□) in vitro plaque forming cell response per culture (PFC/culture) against an antigen. The mean (±SD) PFC response was analyzed in chinook salmon from the Green-Duwamish System and the Nisqually System. The asterisk (*) indicates the secondary PFC/culture that is significantly higher ($P \leq 0.05$) than that observed in the primary response. From Arkoosh et al. 1991.

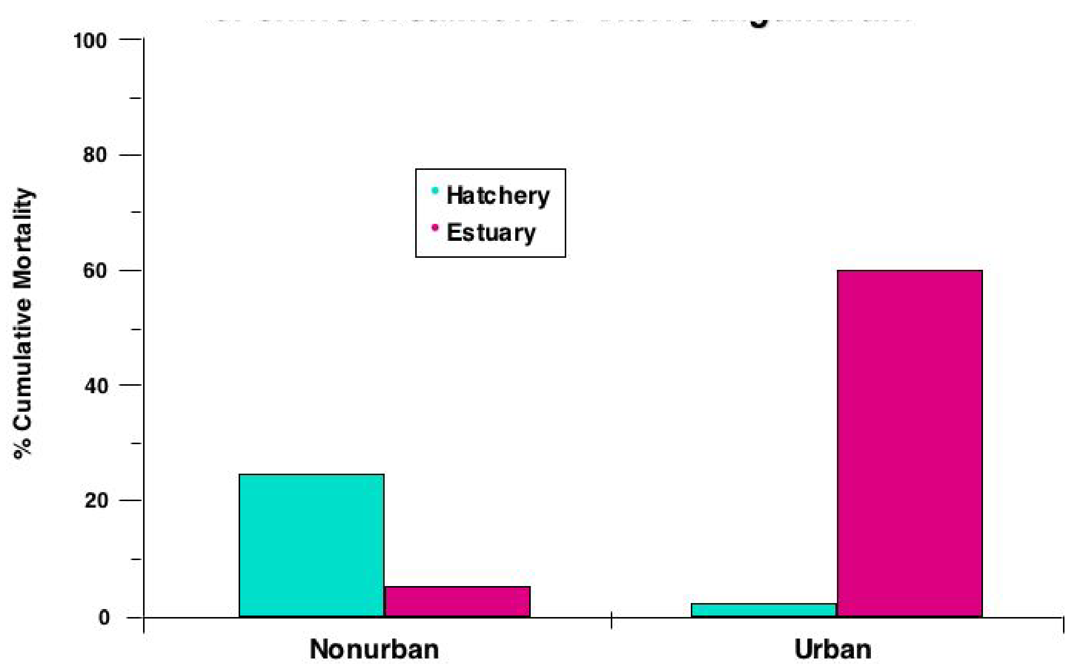

Figure 28. Cumulative mortality of juvenile chinook salmon collected from the Duwamish Waterway (urban) and the associated hatchery hatchery-estuary system and the Nisqually hatchery estuary system (non-urban) following disease challenge with the marine bacterium *Vibrio anguillarum.* Adapted from Arkoosh et al. 1991.

In other studies in Puget Sound, juvenile salmon from urban estuaries (e.g. the Duwamish and Hylebos waterways) had significantly higher levels of DNA damage (i.e., PAH-DNA adducts in liver) than salmon from relatively uncontaminated sites. In addition, salmon from the urban estuaries had significantly higher induction of cytochrome P4501A (CYP1A), the enzyme that metabolizes selected contaminants including PAHs, dioxins and furans, and dioxin-like PCBs (Stein et al. 1995; McCain et al. 1990; Varanasi et al. 1993; Collier et al. 1998a,b; Stehr et al. 2000). These biochemical alterations are not necessarily indicative of adverse health effects in themselves, but are associated with reproductive and developmental abnormalities and liver disease (Williams et al. 1998; Whyte et al. 2000; Myers et al. 2003).

At the two contaminated sites where most of the work described above was done, the Duwamish and Hylebos Waterways, average sediment PCB concentrations ranged from 400-500 ng/g dry wt and average PAH concentrations were about 10,000 ng/g dry wt (Collier et al. 1998a; Krahn et al. 1998). These sediment concentrations are within the range of those reported in the estuary (e.g,. EMAP 2000), although higher than those typically found in Estuary sediments. Total body PCB concentrations in fish collected from the Duwamish and Hylebos sites were in the 250-350 ng/g dry wt range, comparable to some fish sampled from sites within the Estuary.

Various health effects have also been documented in non-salmonid fish and other aquatic biota from the Lower Columbia have, including fish with external abnormalities or skeletal deformities (Markle 1995; Tetra Tech Inc 1995b), alteration in endocrine response function (Goodbred et al. 1997; Foster et al. 2001a,b), and pollution-associated liver lesions (Myers et al. 1994). Although the exposure patterns and life histories of listed salmon may differ from those of these fish species, these data raise additional concern about the potential for exposure and health impacts on salmon.

In addition to bioaccumulative contaminants, waterborne contaminants such as dissolved metals and current use pesticides may pose a threat to listed salmon. Various OPs such as diazinon, carbofuran, and chlorpyriphos at concentrations of 1-10 ug/L can disrupt olfactory function in salmon after exposures of as little a few hours or days (Moore and Waring 1996; Waring and Moore 1997; Scholz et al. 2000; Sandahl et al. 2004). Scholz et al. (2000) reported that the organophosphate pesticide, diazinon, disrupted olfactory function in Chinook salmon at concentrations of 1-10 ug/L, so that fish failed to show normal anti-predator responses or homing behavior (Figure 29).

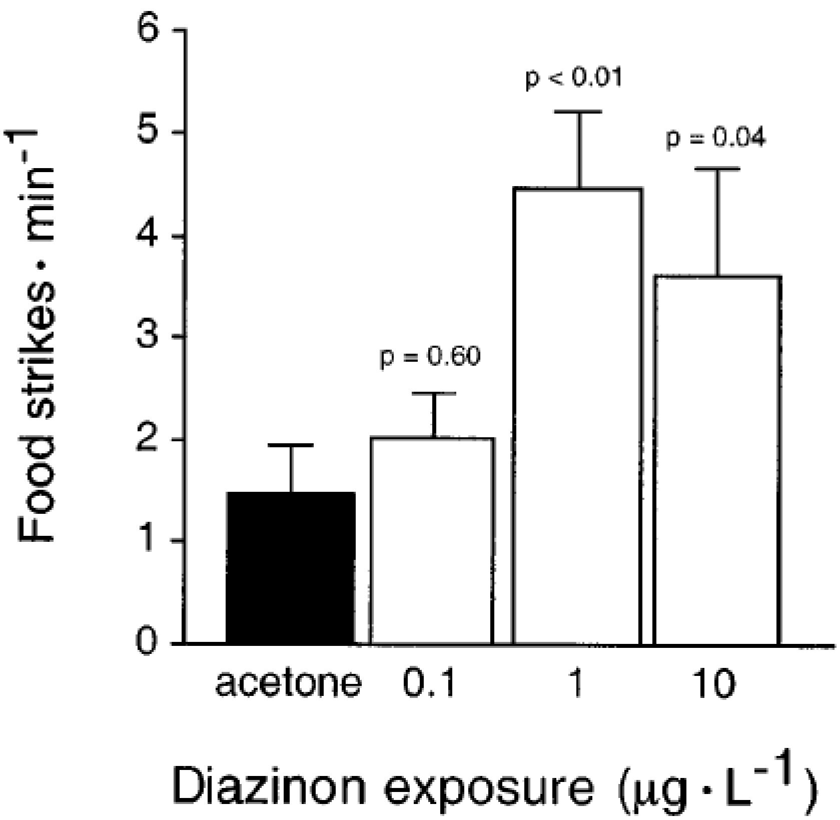

Figure 29. Foraging activity in the presence of olfactory signals of a potential predator in juvenile Chinook salmon exposed to diazinon. Control fish (solid bar) responded to the conspecific skin extracts by reducing their foraging activity and freezing. The magnitude of the antipredator response was reduced in diazinon-exposed fish (2 h at 0.1, 1.0, and 10.0 mg·L^{-1}), and they were more active and fed more often than controls. The effect of diazinon was significant at the 1.0 and 10.0 mg·L^{-1} exposures (P = 0.05, Fisher's test). From Scholz et a. 2000.

Similar responses were observed by Sandahl et al. (2004) with chlorpyrifos at concentrations as low as 0.72 μg/L. Moore and Waring (1996) and Waring and Moore (1997) found that exposure to diazanon and carbofuran in a similar range could desynchronize the reproductive physiology of prespawning Atlantic salmon (*Salmo salar*) by inhibiting the male's ability to detect sex pheromones. Concentrations of diazanon in the 1-10 ug/L range have been reported in NASQAN sampling in the estuary, and other pesticides with related modes of action that would be likely to have the same effects (e.g., chlorpyriphos, malathion, aldicarb, carbaryl, carbofuran) are detected even more frequently and at higher concentrations.

Similarly, Baldwin et al. (2003) and Sandahl et al. (2004) found that exposure to copper at concentrations in the 3-6 μg/L range for as little as 30 min affected olfactory function in coho salmon so they could no longer respond normally to test odorants (Figure 30). This could impair the ability of juveniles to find prey and avoid predators, or interfere with homing and reproductive behavior in adults. Dissolved copper concentrations at the estuary sites sampled in the USGS NAQAN survey were within this range (Fuhrer et al. 1996), and copper in suspended sediments was substantially higher (45-120 μg/L). Other contaminants in the water column, including endocrine-disrupting substances such as synthetic hormones, are only beginning to be characterized in the in this part of the basin, but potentially could also have substantial impacts on salmon.

More research is clearly needed to document exposure and associated effects of chemical contaminants on endangered Columbia Estuary salmon, but the available data show that environmental concentrations and tissue burdens of several classes of contaminants are within the range where they could primarily affect abundance and population growth rate in listed stocks. The true magnitude of the effect is uncertain, but a recent modeling study suggests it could be significant for at least some ESUs. Spromberg and Meador (2004) used life cycle models to examine the impacts of low-level toxic effects (10-25% response level for mortality, immune suppression, and growth) on the population dynamics of fall run chinook salmon.

Responses in this general range might be expected in estuary fish exceeding, for example, the tissue benchmark for PCBs developed by Meador (2002). These results indicate that after 20 years of continued reductions at the 10% level, population abundance was severely depressed (up to 2-3 times lower than non impacted populations) for several of the endpoints. When the 25% toxicity response was modeled for 20 years, population abundance was between 3 and 20 times lower, depending on the endpoint.

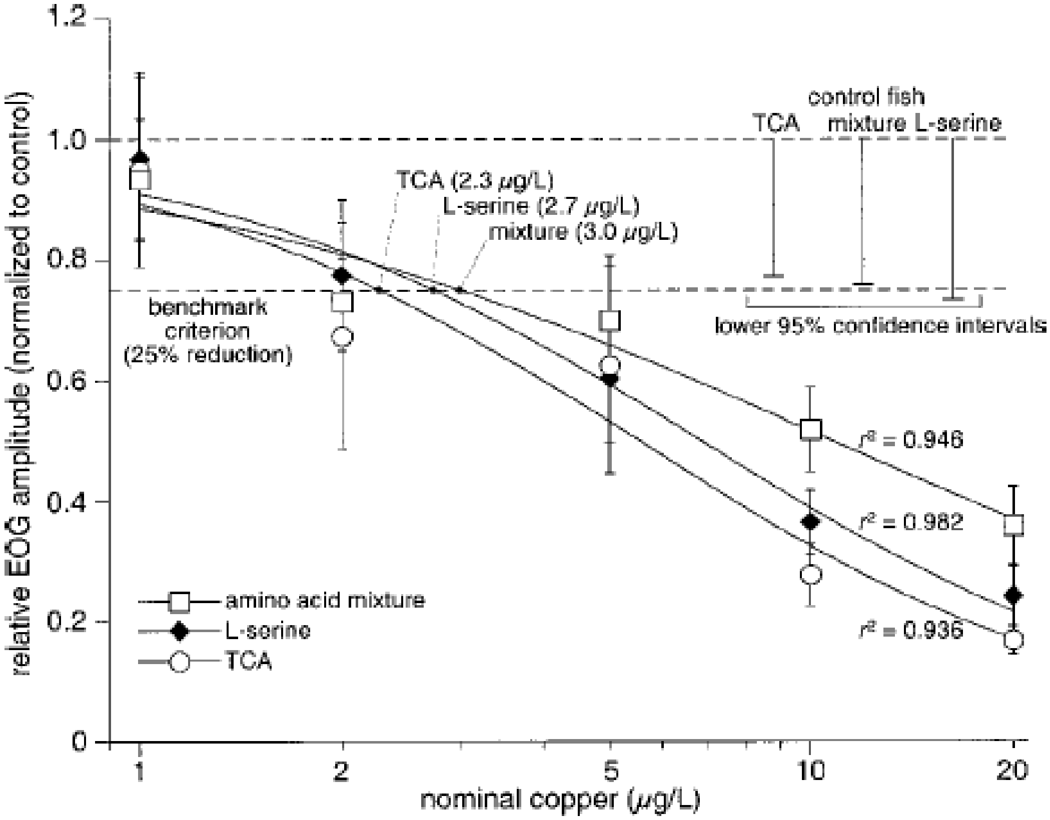

Figure 30. Dose–response curves and threshold determinations for sublethal copper neurotoxicity. Data were obtained from six treatment groups (control and five copper exposures; n = 6 fish per group). The evoked electro-olfactogram (EOG) amplitudes for all copper exposures were normalized to the mean response of the controls and expressed as a group (mean ± 1 standard error). The dashed line indicates a benchmark criterion of 0.75, or a 25% reduction in evoked response (relative to controls). Three vertical lines in the upper right show the lower limits of the 95% confidence interval for the control response to the three different odorants. Filled circles indicate the benchmark concentrations for the different olfactory pathways (Lserine, taurocholic acid [TCA], and the amino acid mixture). Note that the benchmark values are nominal concentrations, or a change (increase) from the copper present at approximately 3 mg/L in the source water for the Northwest Fisheries Science Center hatchery (Seattle, WA, USA). From Baldwin et al. 2003.

In summary, exposure to chemical contaminants has the potential to affect survival and productivity of both ocean and stream-type stocks in the estuary. Stream-type ESUs are most likely to be affected by short-term exposure to waterborne contaminants such as current use pesticides and dissolved metals that may disrupt olfactory function and interfere with associated behaviors, such as capturing prey, avoiding predators, imprinting and homing. Ocean-type ESUs will also be exposed to these types of contaminants, but will also be affected by persistent, bioaccumulative toxicants such as PCBs and DDTs, which they may absorb during their more extended estuarine residence. Consequently, it is likely that the impact on ESUs exhibiting the ocean life history type is greater.

Caspian Tern Predation of Juvenile Salmon

In the preceding analyses, we suggested that changes in estuarine habitats affects population viability but that quantifying these changes is problematic. One factor that can be quantitatively linked to VSP criteria is predation. Estimates of life stage specific predation mortality can be directly linked to changes in population growth rate or productivity. In general, many predator prey interactions in the Columbia River estuary have potentially changed from historic conditions including predation by marine mammals (e.g., California sea lions), smallmouth bass *Micropterus salmoides*, northern pikeminnow *Ptychocheilus oregonensis*, and cormorants. For most of these predators, we have little quantitative data on how this has affected population viability.

One exception to this is Caspian Tern *Sterna caspia*. Increasing populations of terns nesting on islands in the Columbia River estuary annually consume large numbers of migrating juvenile salmonids (Roby et al. 1998) and thus constitute one of the factors that may currently limit salmonid stock recovery (Roby et al. 1998; Independent Multidisciplinary Science Team 1998; Johnson et al. 1999). Another exception is northern pikeminnow. Predation by this species has been extensively studied and can be significant source of loss of juvenile salmon below Bonneville Dam.

In this section, we consider mortality of juvenile salmon as a result of predation by Caspian Terns as a limiting factor. In the last section of this report, we examine predation by northern pikeminnow. We did not consider northern pikeminnow as a factor for analysis in this section because of some key information gaps that we were unable to address (e.g., historic levels of predation compared to present levels).

Caspian terns are highly migratory and are cosmopolitan in distribution (Harrison 1983; Harrison 1984). Nesting has been reported throughout North America and in Australia, New Zealand, South Africa, Asia, and Europe. The numbers of Caspian terns in western North America more than doubled between 1980 and 1999 (Cuthbert and Wires 1999). One reason for the increase is that human-created habitat provides high quality nest sites and is associated with population increases in many parts of North America (Cuthbert and Wires 1999).

In the early 1990s, a substantial increase in the size of a newly established Caspian tern nesting colonies on man-made islands in the Columbia River estuary was noted by NOAA Fisheries staff. Several estuary islands on which piscivorous birds nest were created from or augmented by materials dredged to maintain the Columbia River Federal Navigation Channel. Before 1984, there were no recorded observations of terns nesting in the Columbia River estuary, when approximately 1,000 pairs apparently moved from Willapa Bay to nest on newly deposited dredge material on East Sand Island. In 1986, those birds moved to Rice Island. The Caspian tern colonies in the estuary have since expanded to 9,000-10,000 pairs, the largest ever reported. In 1999, the colony was encouraged to relocate to East Sand Island. In 2001, the majority of the West Coast population of terns nested on just four acres on East Sand Island, and in 2002, the terns nested on six acres.

Caspian terns arrive in the Columbia River estuary in April and begin nesting at the end of the month (Roby et al. 1998). To avoid mammal and avian predators, terns construct their nests on islands (Harrison 1984) and show a preference for barren sand. They are piscivorous in nature (Harrison 1984), requiring about 220 grams (roughly one-third of their body weight) of fish per day during the nesting season. The timing of courtship, nesting and chick rearing corresponds with the outmigration of many of the salmonid stocks in the basin (Collis et al. 2002).

Salmon and steelhead constitute a major portion of tern diets, particularly when the birds nested on Rice Island. Diet analyses indicated that juvenile salmonids constituted 77.1% of prey items in 1997 and 72.7% of prey items of Caspian terns nesting on Rice Island (Collis et al. 2002). During the May peak of smolt out-migration of steelhead, yearling chinook salmon, and coho salmon through the estuary, when Caspian terns are in their incubation period, the diet of Caspian terns was consistently over 90% juvenile salmonids (Collis et al.2002). This concentration on salmon as a food source translates into substantial juvenile mortality during the outmigration period.

Efforts to relocate the terns to East Sand Island from Rice Island that began in 1999 have succeeded in reducing consumption of smolts without affecting tern productivity. East Sand Island is a site lower in the estuary with abundant alternate prey sources. Over the last few years, studies suggest that consumption of salmonids in the estuary has been lower than previous levels while consumption of alternative prey species has increased. Relocating the colony to East Sand Island, which is lower in the estuary and closer to periodically abundant Pacific herring *Clupea harengus pallasi* and anchovies *Engraulis mordax*, has contributed to the reduction. Observed diets, which consisted of almost exclusively salmonids at Rice Island (77% in 1999 and 90% in 2000), shifted to 46%, 47% and 33% salmonids at East Sand Island in 1999, 2000 and 2001 respectively (Collis et al. 2001a; Roby et al. 2003).

These data suggest that substantial declines in juvenile salmonid mortalities from Caspian tern predation. This is supported by studies estimating tern consumption. In 2000, salmonid consumption by terns was estimated at 7.3 million smolts, which is 4.4 million less than in 1999--the last time a substantial number of terns nested on Rice Island (Collis et al. 2001a; USFWS 2001). In 2001, salmonid consumption was estimated at 5.9 million smolts, which is 5.9 million less than in 1999 (Collis et al. 2001a). These data were substantiated by PIT (Passive Integrated Transponders) tag detections on the two islands in 1999 and 2002. Approximately 2 to 4 times fewer tags per pair of terns were detected per nest on East Sand Island in 1999 and 2000 than were detected on Rice Island in 1999 and 2000.

In a recent evaluation of the impact of Caspian tern predation on juvenile salmon (Good et al. 2003), two approaches were recognized as providing the types of predation rate estimates that are needed for salmon life cycle models that are used to assess the effects of various factors on risk of extinction in the Columbia River basin (Kareiva et al. 2001). One approach uses bioenergetics modeling. Since 1997, biologists with the Bonneville Power Administration- funded research project, *Avian Predation on Juvenile Salmonids in the Lower Columbia Rive*r, a joint project of Oregon State University, the U.S. Geological Survey, the Columbia River Inter-Tribal Fish Commission, and Real Time Research Consultants) have used observed salmonid consumption at tern colonies in a bioenergetics model (Roby et al. 1998) to estimate the consumption of salmonids in the Columbia River estuary. S molt consumption estimates from 1999 to 2002 using this approach ranged from a low of 5.9 to a high of 11.7 million smolts eaten.

Another approach uses detections of passive integrated transponders (PIT) tags on Caspian tern colonies to estimate salmonid predation rates overall as well as by ESU (Collis et al. 2001b, Ryan et al. 2001). Since 1987, researchers in the Columbia River basin have placed over five million PIT tags in juvenile salmonids for a variety of studies (Ryan et al. 2001). Identifying PIT tags on bird colonies can provide a minimum estimate of proportion of the stocks that were consumed by terns in these colonies. In recent years, approximately one million juvenile salmonids have been PIT-tagged annually (Collis et al. 2001b), the vast majority of which are steelhead and chinook from the Snake River basin. Using PIT tag detection equipment, over 115,000 PIT tags were detected on Rice Island in 1998 and 1999 (Ryan et al. 2001).

Of the PIT tags placed in steelhead smolts in 1997 that were detected at Bonneville Dam, 2.8% of wild smolts and 5.4% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b). For steelhead that were PIT-tagged in 1998 and detected at Bonneville Dam, 11.7% of wild smolts and 13.4% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b). For yearling chinook salmon that were PIT tagged in 1998 and detected at Bonneville Dam, 0.5% of wild smolts and 1.6% of hatchery-raised smolts were subsequently detected on the Rice Island tern colony (Collis et al. 2001b).

Ryan et al. (2003) analyzed PIT tag data from 1998 to 2000 on Rice Island and East Sand Island and determined that steelhead experienced higher predation rates (0.6% to 8.1% on East Sand Island and 1.3% to 9.4% on Rice Island) than chinook salmon (0.2% to 2.0% on East Sand Island and 0.6% to 1.6% on Rice Island). Overall, Caspian terns consumed approximately 6% to 14% of the estimated outmigrating population of juvenile salmonids originating from the Columbia River basin.

In a recent analysis of the impact of Caspian tern predation on salmon recovery, efforts focused on determining if a unique predation rate could be identified. The effort focused on the Caspian tern colonies on East Sand Island in the lower estuary of the Columbia River because the colony currently represents the majority of the West Coast Caspian tern population. The focus period was 1999-2002 because this represents the time period after relocation from Rice Island during which this colony has dominated Caspian tern predation activity in the Columbia River estuary. Bioenergetics modeling was used to calculate predation rates (estimated number of salmon consumed/estimated number of salmon available in the estuary) using updated and refined estimates of the number of outmigrating salmon that migrate through the river or are transported and released below Bonneville Dam. PIT tag detections were also used to generate estimates of predation rates on salmon.

Although the relationship between tern abundance and predation rate is not known with certainty, the estimates (using either bioenergetics modeling or PIT tag data) showed a linear relation between predation rate on all salmon to the number of Caspian terns nesting on East Sand Island during the breeding seasons of 1999-2002 (Figure 31). Moreover, PIT tag detection also allows ESU-specific predation rate estimates to be derived. Support for a linear relationship between estimates of predation rate and the number of terns nesting on East Sand Island comes from per capita consumption rates (# of smolts consumed/adult tern), which have been relatively constant throughout the range of colony sizes on East Sand Island from 1999-2003. The per capita consumption rate in 1999 (mean = 437.5) was nearly equivalent to that of 2000 (mean = 431.1), even though there was an almost five-fold difference in colony size (Figure 32).

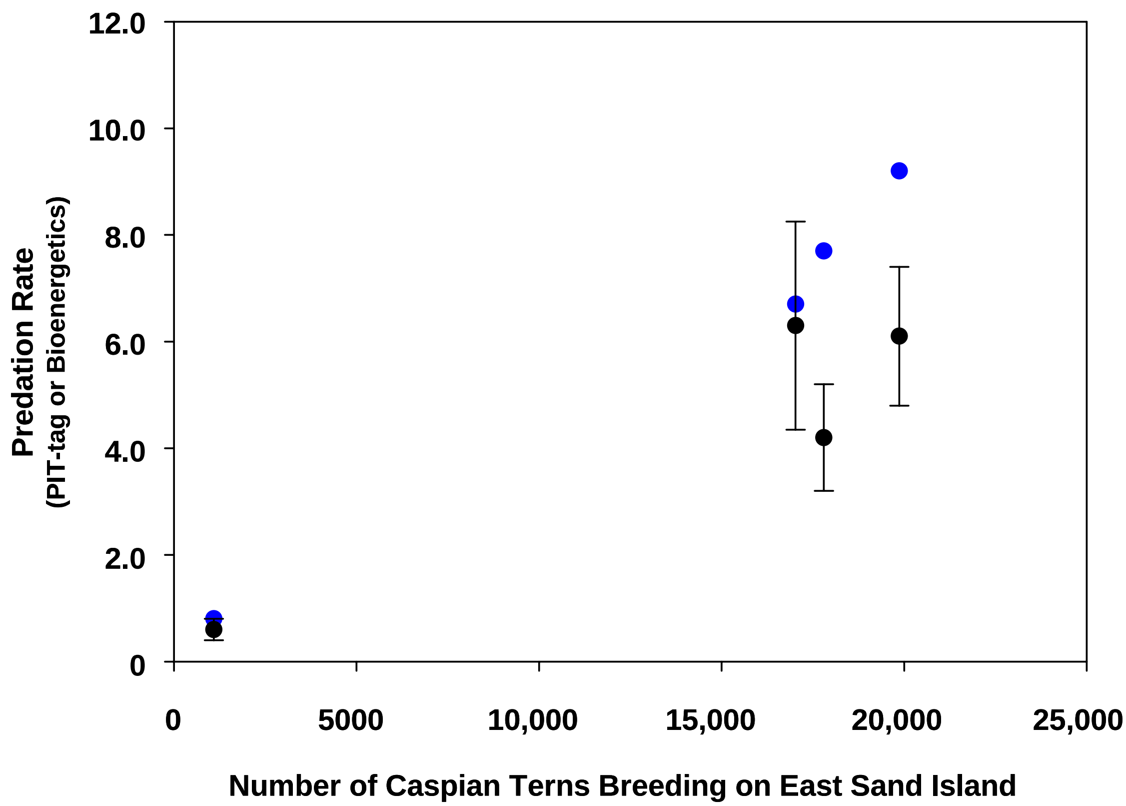

Figure 31. Estimated predation rates on all steelhead in the Columbia River estuary by Caspian Terns (1999-2002) using bioenergetics modeling (black symbols) and recovery of PIT tags (blue symbols). Error bars on bioenergetics estimates represent 95% confidence limits.

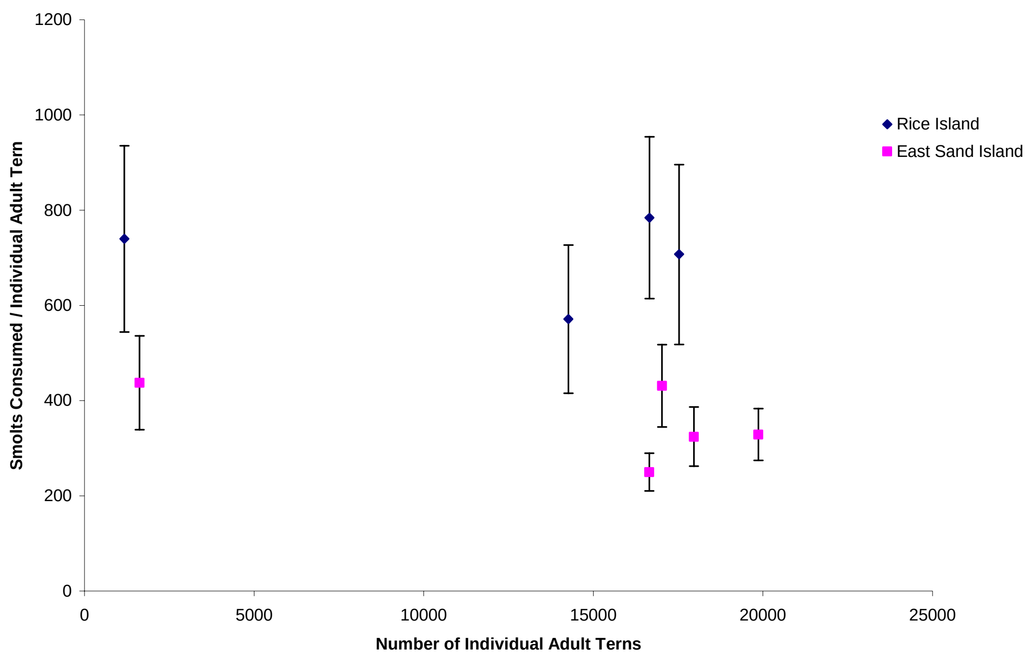

Figure 32. Per Capita Smolt Consumption by Columbia River Estuary Caspian Terns on all Salmonids 1997-2003 (with 95% CI). From D. Lyons and D. Roby. Oregon Fish Cooperative Unit, Oregon State University. (Pers. Comm.)

NOAA Fisheries has developed a life cycle model under the auspices of the Cumulative Risk Initiative (CRI) at the Northwest Fisheries Science Center to assess salmonid population trends and the impact of an anthropogenic activity on those trends. This model has application when mortality rates can be constructed and attributed to a particular source. The value of life cycle models derive from providing an objective outcome for comparing the influence of various factors influencing population growth rates, rather than attempting to estimate real gains from any management action. Assessing the impact of predation by Caspian terns on juvenile salmonids during a particular life history phase was amenable to such evaluation.

Using the CRI model, Good et al. (2003) estimated the impact of Caspian tern predation on the population growth rate (λ) of all steelhead and Spring Chinook salmon in the basin using predation rate estimates derived from bioenergetics modeling and PIT tag detections. Because of the similarity in the results between the two approaches, we present information only from estimates derived from PIT tag detections, as ESU specific impacts can also be derived with this information.

The predation rate for 20,000 Caspian terns on all steelhead and spring Chinook salmon was estimated using the regression equations generated from PIT tag detections. This number of terns represents the maximum number observed to date on East Sand Island. Reductions in predation rate corresponding to reduced tern population sizes were used to model the potential increase in λ, assuming all steelhead or spring Chinook salmon mortality attributable to terns is not compensated for by mortality due to other sources. The maximum proportional increase in λ corresponding to complete elimination of mortality due to tern predation (i.e. removal of all terns from the estuary) was 1.9% and 0.8% for steelhead and spring Chinook salmon, respectively, using the PIT-tag estimate of predation rate.

The PIT tag detection approach enables the calculation of ESU-specific estimates of predation rate (and hence proportion increase in λ). Good et al (2003) used the CRI model to estimate impact of Caspian tern predation on the population growth rate (λ) of steelhead and spring Chinook salmon ESUs using predation rates estimated from PIT tag detections for which reasonable estimates of the number of smolts available to be consumed could be generated. Predation rates for 20,000 Caspian terns on four of the five ESA-listed steelhead and spring Chinook salmon ESUs were estimated using linear regression. The maximum proportional increase in λ corresponding to complete elimination of mortality due to tern predation ranged from 1.9% to 4.9% for steelhead ESUs.

Several factors must be considered when interpreting the results of these calculations. Perhaps the most important is that this type of calculation assumes that there is no compensatory mortality later in the life cycle, and that any reduction in tern predation is fully realized. In their assessment of predation impact by Rice Island terns on salmonids in 1997-1998, Roby et al (2003) hypothesized that tern predation was 50% additive. Given these limitations and uncertainties, the estimates of percent change in population growth rates should be viewed as maximum potential improvements. Realized improvements in population growth would likely be lower from any management action that reduces Caspian tern predation impacts on salmonid ESUs.

These results may not be as easy to achieve as they are to calculate. It is also important to recognize that other factors such as ocean conditions may also influence population growth rate to a greater degree than the potential gains that may be realized from reducing predation by one species of avian predator on one island located in the lower estuary of the Columbia River basin.

Overall, it is evident that Caspian tern predation affects primarily salmon and steelhead that exhibit a stream type life history rather than an ocean type life history as they move and utilize the Columbia River estuary. This is primarily a result of the dominant migratory periods employed by salmonids with a stream type life history. Salmon from this life history type move in great numbers at the time Caspian terns begin nesting (May through June) and have the greatest energetic needs for chick production.

Although there are some impacts to juvenile salmon exhibiting an ocean type life history, characteristic of fall Chinook in the Columbia River basin, the impact is less than for the stream type salmonids (Roby et al. 2003),. Good et al. (2003) concluded that gains in λ for steelhead ESUs were comparable to gains that could be derived from additional improvements to the FCRPS to increase survival, but much less than can be achieved by harvest modifications. Because steelhead ESUs were most strongly affected by Caspian tern predation, improvements to λ by managing terns were considered to benefit other salmon ESUs in the basin, albeit to a much lesser degree.

THE EFFECTS OF ESTUARINE FACTORS ON RECOVERY POTENTIAL OF ESUs

In this section, ESUs, life history type, life history strategy, limiting factor impacts, habitat attributes, and population viability are integrated in order to compare and contrast the importance of each limiting factor on population viability. We consider all listed ESUs in the Columbia River Basin in addition to Lower Columbia River coho salmon because of the possibility that this ESU will get listed in the future. The focus of this analysis was on effects of estuarine factors on population viability, not on the relative importance of the estuary relative to other factors operating at other life stages. This broader analysis is outside the scope of this review and will be considered in the Integration Phase of the overall life cycle analysis.

We evaluated effects of factors within two zones of the estuary: Bonneville Dam to the mouth was defined as one zone and the plume was considered a second major zone. Within the Bonneville to the mouth zone, we also differentiated shallow, low velocity, vegetated habitats from medium and deep, higher velocity habitats generally associated with the main channels. We were limited to this broad scale type of analysis because a finer scale evaluation of how different juvenile life history strategies use habitats and zones is not possible with existing information.

For example, we know fry and fingerling strategies are more closely associated with shallow, low velocity habitats (e.g., swamps, emergent marshes, and shallow flats) and less associated with medium and deep, higher velocity channel habitats while the opposite pattern exists for larger size classes such as yearlings (Healey 1980, 1982; Levy and Northcote 1981, 1982; Simenstad et al. 1982; Levings et al. 1986; Miller and Sadro 2003). But we do not know how different types of shallow water habitats are used (e.g., emergent marsh vs swamp vs mudflat) within a zone. While differences are likely in use of different habitats between zones, we lacked the ability to make this type of discrimination other than between the confined portion of the estuary vs the plume.

To guide our evaluation, we used the following hypotheses or guiding principles about use of estuarine habitats by juvenile salmon and the effects of specific limiting factors that were developed in previous sections:

1. Tern predation differentially affects the larger yearling strategies, especially steelhead, more than smaller life history strategies such as fingerling chinook (Ryan et al. 2003).

Tern predation is assumed to occur in the estuary zone but primarily in medium and deep water channel habitat rather than shallow water areas (Tables 5,6). Tern predation is assumed to be minimal in the plume.

2. The main effect of flow reductions is to affect amount of shallow water habitat available to fish and opportunity for the fish to use the habitat; the main effect of habitat changes is on distribution, quantity and quality of habitat; the main effect of toxics is on habitat quality (capacity).

3. Any reduction in quality or quantity of shallow water habitat affects smaller juvenile salmonids employing strategies such as fry and fingerlings significantly more than subyearlings and yearlings (Table 5, 6). From the perspective of ocean type populations in the estuary, changes in the quantity and quality of shallow water habitats most impacts viability of these populations.

4. Subyearling and yearlings primarily use medium and deep channel habitat.

5. Fry and early fingerling life history strategies do not move into the plume, but more likely utilize the surf zone when they exit the estuary proper.

6. Reductions in flow above Bonneville affect the size and shape of the plume. Primarily as a result of flow but also know doubt also a result of physical changes to the estuary (eg., dredging and diking), the shape, behavior, size, and composition of the plume has been changed.

7. Toxics impact the quality of habitat but consequences of toxics can occur downstream of where the burden was acquired. The impact, though, is assumed to be associated with the habitat where the exposure occurs.

8. Flow and habitat changes in the estuary are interrelated.

9. Cumulative impacts were not considered in the analysis.

Table 5. Linkages between limiting factors associated with the estuary upstream of the mouth, life history strategies, life history type (ocean type and stream-type) and ESU (for a complete list see Table 3). Only two, general habitat types in the estuary were considered- shallow, low velocity and medium/deep, channel higher velocity. Factors were ranked as having a high, medium, or low ability to affect the relative abundance of particular life history strategies.

ESU*	Life history type	Life history strategy	Shallow, low velocity				Deep, channel			
	Ocean-Type		Flow	Habitat	Terns	Toxics	Flow	Habitat	Terns	Toxics
		Early fry	High	High	Low	Medium	Low	Low	Low	Low
SR		Late fry	High	High	Low	Medium	Low	Low	Low	Low
Fall,		Early fingerling	High	High	Low	Medium	Low	Low	Low	Low
LCRC,		Late fingerling	High	High	Low	Medium	Low	Low	Low	Low
		Subyearling	Low	Low	Medium	Low	Medium	Low	Medium	Low
		Yearling	Low	Low	Medium	Low	Medium	Low	Medium	Low
LCRS,	Stream-Type	Early fry	Medium	Medium	Low	Low	Low	Low	Low	Low
UCRS		Late fry	Medium	Medium	Low	Low	Low	Low	Low	Low
C		Early fingerling	Medium	Medium	Low	Low	Low	Low	Medium	Low
		Late fingerling	Medium	Medium	Low	Low	Low	Low	Medium	Low
		Subyearling	Low	Low	Medium	Medium	Medium	Low	Medium	Low
		Yearling	Low	Low	Medium	Medium	Medium	Low	Medium	Low

* SR Fall= Snake River Fall chinook salmon, LCRC= Lower Columbia River chum salmon, LCRS= Lower Columbia River steelhead, UCRSC= upper Columbia River spring chinook.

Table 6. Linkages between limiting factors associated with the plume, life history strategies, life history type (ocean type and stream-type) and ESU (for a complete list see Table 3). The plume was only considered as one habitat zone. Factors were ranked as having a high, medium, or low ability to affect the relative abundance of particular life history strategies using the plume.

ESU[1]	Life History Type	Life History Strategy	Plume			
	Ocean-Type		Flow	Habitat	Terns	Toxics
		Early fry	Low	Low	Low	Low
SR		Late fry	Low	Low	Low	Low
Fall,		Early fingerling	Medium	Low	Low	Low
LCRC,		Late fingerling	Medium	Low	Low	Low
		Subyearling	Medium	Low	Medium	Low
		Yearling	Medium	Low	Medium	Low
LCRS,	Stream-Type	Early fry	Low	Low	Low	Low
UCRS		Late fry	Low	Low	Low	Low
C		Early fingerling	Medium	Low	Medium	Low
		Late fingerling	Medium	Low	Medium	Low
		Subyearling	Medium	Low	Medium	Low
		Yearling	Medium	Low	Medium	Low

1- SR Fall= Snake River Fall chinook salmon, LCRC= Lower Columbia River chum salmon, LCRS= Lower Columbia River steelhead, UCRSC= upper Columbia River spring chinook.

For stream-type ESUs (e.g., Snake River spring/summer Chinook salmon and mid Columbia River steelhead), the primary estuarine factors affecting population viability are tern predation and flow (Tables 7-8 and 10). Tern predation was ranked in the medium category for several reasons. First, tern predation is primarily directed at subyearling and yearling size fish which are the dominant strategies in stream type ESUs such as Snake River steelhead. Second, these larger fish occur in habitats (deeper water channel habitats) where they are most vulnerable to the terns. Third, tern predation significantly affects abundance and productivity; scores for these parameters were doubled if we assumed there was an affect. Based upon anecdotal observations of NOAA Fisheries working in the Columbia River plume, we assumed that most tern predation occurred upstream of the river mouth. If significant predation did occur in the plume, then the score for this factor would increase.

Flow changes were also ranked medium for stream type ESUs because both abundance and productivity were affected and the main life history strategies for these ESUs are most vulnerable to predators in the plume habitat. Toxics and habitat were ranked low for stream type ESUs because the main life history strategies associated with this ESU do not occupy the habitat where the main affects occurs.

For ocean type ESUs (Lower Columbia River Fall Chinook Salmon and Lower Columbia River chum), flow and habitat were rated as having a high ability to affect population viability (Tables 7, 9, and 11). Flow and habitat affects are most significant in shallow water areas; both the quantity of habitat and the opportunity to use this habitat are affected. Finally, both the flow and habitat limiting factors affect all VSP parameters for ocean type populations. The loss of shallow water habitat and changes in its distribution and quality caused by flow and habitat changes will reduce the capacity of estuarine habitats to support ocean type populations; this will reduce abundance and productivity of these populations. Further, loss and degradation of shallow water habitat will also diminish the spatial structure and number of life history pathways available to the fish. This has the potential to make these populations more vulnerable to effects of extreme events such as severe droughts or strong El Nino events. As we have noted, the losses of shallow water habitats due to the combined effects of flow and habitat changes are dramatic while losses of deeper water habitats appear to be minimal.

Table 7. Summary rating table for listed Columbia River Basin ESUs for estuary factors. Ranks were assigned based upon the following ranges: low (0-0.32), medium (0.33-0.66) and high (0.67-1.00).

Life History Type	Stream Type				Ocean Type			
ESUs	Snake River Spring/Summer Chinook Upper Columbia River Chinook Snake River Steelhead Upper Columbia River Steelhead Middle Columbia River Steelhead Lower Columbia River Steelhead Upper Willamette Steelhead Upper Snake River Sockeye Lower Columbia River Coho				Lower CR Chum Salmon Snake River Fall Chinook Upper Willamette Chinook Lower Columbia River Fall Chinook			
Rating Level	Factor				Factor			
	Tern Predation	Toxics	Habitat	Flow	Tern Predation	Toxics	Habitat	Flow
Level 1	6	5	4	7	3	5	8	7
Level 2	6	2	0	3	2	6	6	7
TOTAL SCORE	12	7	4	10	5	11	14	14
TOTAL POSSIBLE	20	28	20	20	20	28	20	20
RATIO	**0.60**	**0.25**	**0.20**	**0.50**	**0.25**	**0.39**	**0.70**	**0.70**
RANK	**Medium**	**Low**	**Low**	**Medium**	**Low**	**Medium**	**High**	**High**

Table 8. Level 1 ratings of estuary factors for stream-type ESUs (Snake River spring/summer chinook, Upper Columbia River chinook, Snake River Steelhead, Upper Columbia River Steelhead, Middle Columbia River Steelhead, Lower Columbia River steelhead, Upper Willamette Steelhead, Upper Snake River sockeye, and Lower Columbia River coho). An answer to a question of yes equals a 1 other than for the VSP criteria of productivity and abundance which are scored a 2 for yes. An answer of no equals a 0.

Screening Criteria	Factor			
	Tern Predation	Toxics	Habitat	Flow
LEVEL 1- IS THE FACTOR OF CONCERN FOR THE ESU?				
What is the relevance of the factor to the ESU?				
Are there large numbers of fish affected (2x)	2	2		2
Is there a significant affect on productivity (2x)	2	2		2
Is there a significant affect on LH Diversity			1	1
Is there a significant affect on spatial structure			1	1
What is the level of change possible in factor?				
Is there a significant change from historic levels	1	1	1	1
Is the amount of Improvement possible substantial	1		1	
Score	6	5	4	7
Max Possible Score	8	8	8	8

Table 9. Level 1 ratings of estuary factors for ocean-type ESUs (Lower Columbia River chum salmon, Upper Willamette Chinook, Lower Columbia River fall chinook, and Snake River fall chinook). An answer to a question of yes equals a 1 other than productivity and abundance which are scored a 2 for a yes. An answer of no equals a 0.

Screening Criteria	Factor			
	Tern Predation	Toxics	Habitat	Flow
LEVEL 1- IS THE FACTOR OF CONCERN FOR THE ESU?				
What is the relevance of the factor to the ESU?				
Are there large numbers of fish affected (2x)		2	2	2
Is there a significant affect on productivity (2x)		2	2	2
Is there a significant affect on LH Diversity	1		1	1
Is there a significant affect on spatial structure			1	1
What is the level of change possible in factor?				
Is there a significant change from historic levels	1	1	1	1
Is the amount of Improvement possible substantial	1		1	
Score	3	5	8	7
Max Possible Score	8	8	8	8

Table 10. Level 2 ratings of estuary factors for stream-type ESUs (Snake River spring/summer chinook, Upper Columbia River chinook, Snake River Steelhead, Upper Columbia River Steelhead, Middle Columbia River Steelhead, Lower Columbia River steelhead, Upper Willamette Steelhead, Upper Snake River sockeye, and Lower Columbia River coho). With the exception of toxics (see footnote), an answer to a question with a yes equals a 1. An answer of no equals a 0.

Screening Criteria	Terns			Toxics[a]			Habitat			Flow		
	SW[b]	DW	Pl	SW	DW	Pl	SW	DW	Pl	SW	DW	Pl
LEVEL 2- SIGNIFICANCE OF FACTOR												
For the dominate LHS, is the relative impact on numbers by habitat type significant?	1	1	1		1							1
For the dominate LHS, does the factor significantly affect habitat--												
1. Quantity												
2. Quality					1							1
3. Opportunity	1	1	1									1
Score	2	2	2	0	2	0	0	0	0	0	0	3
Total Factor Score		6			2			0			3	
Max Possible Score		12			20			12			12	

a- Scores for toxics include a value for sediment and water in estuary (ie, the sw quality score can be a 2) and water in the plume.

b- SW=Shallow water from the river mouth to Bonneville, DW=Deep water from the river mouth to Bonneville, Pl=Plume

Table 11. Level 2 ratings of estuary factors for ocean-type ESUs (Lower Columbia River chum salmon, Upper Willamette Chinook, Lower Columbia River fall chinook, and Snake River fall chinook). With the exception of toxics (see footnote), an answer to a question with a yes equals a 1. An answer of no equals a 0.

Screening Criteria	Terns			Toxics[a]			Habitat			Flow		
	SW[b]	DW	Pl	SW	DW	Pl	SW	DW	Pl	SW	DW	Pl
LEVEL 2- SIGNIFICANCE OF FACTOR												
For the dominate LHS, is the relative impact on numbers significant by habitat type?	1			2	1		1			1		1
For the dominate LHS, does the factor significantly affect habitat--												
1. Quantity							1		1	1		1
2. Quality	1			2	1		1		1	1		
3. Opportunity							1			1		1
Score	2	0	0	4	2	0	4	0	2	4	0	3
Total Factor Score		2			6			6			7	
Max Possible Score		12			20			12			12	

a- Scores for toxics include a value for both sediment and water in estuary (ie, the sw quality score can be a 2) and water only in the plume.

b- SW=Shallow water from the river mouth to Bonneville, DW=Deep water from the river mouth to Bonneville, Pl=Plume

Flow and habitat also have a significant affect on ocean type populations because these populations are dominated by small size classes (fry and fingerlings), make extensive use of shallow water habitats, and have the longest residence time in the estuary (i.e., they are most dependent upon shallow water habitats) (e.g., Bottom et al. 2001). A major function of these shallow water habitats for these small size classes is to support their feeding and growth; high growth rates experienced here can help population members avoid some of the predation that these small fish experience (Simenstad et al. 1982). Yearling and subyearling fish are generally not in the habitats where they are most vulnerable to the effects of these two factors. Because relatively few stream type fish use shallow water habitats, we predict that flow and habitat will have less of an effect on capacity of these habitats to rear and support stream type populations but a more significant affect on diversity and spatial structure of these populations (Table 7). Because of the loss of shallow water, estuarine dependent strategies (i.e., fry and fingerlings), the number and quality of the spatial and temporal trajectories expressed by these populations will decline.

Effects of toxic contamination on ocean type ESUs was rated medium. Both water borne and sediment contaminants can affect these life history strategies in shallow water areas where the dominant life history strategies are most abundant. We assumed that toxics impact the quality of habitat upstream of the river mouth and that there was not significant affects in the plume. The consequences of the uptake of toxics can occur downstream of where the burden was acquired including if the exposure occurred above Bonneville. However, we assumed the impact was associated with the habitat where the exposure occurred. Tern predation has a low affect on this ESU because terns do not target fry and fingerling strategies (the dominant ones associated with this ESU).

OTHER CONSIDERATIONS IN EVALUATING THE ROLE OF THE ESTUARY IN RECOVERY OF ANADROMOUS SALMONIDS

Data Gaps

While our intent in this report was not to define all major information needs (this is done more comprehensively by Bottom et al. 2001), there were several data gaps that notably constrained our ability to analyze effects of factors on population viability which are important to highlight. One of these is clearly the lack of information on juvenile salmon use of the estuary by different populations, especially understanding how salmon use different geographic zones and different habitats within each zone. Because of this lack of habitat and zone specific data, we had to assume that all shallow water areas are similarly used between the mouth and Bonneville Dam.

A second data gap was the lack of knowledge regarding adult salmon use of the estuary. Our entire analysis was based upon juvenile salmon use of the estuary. Although it seems reasonable to hypothesize that some of the factors we considered could impact adult salmon in the estuary, we had no information on adult salmon and their use of estuarine habitats. For example, adult salmon are probably exposed to toxics in the estuary with unknown effects while changes in flow attributes may alter the timing of adult salmon migrations and expose them to predators such as marine mammals for longer periods. The lack of information on adult salmon use of the estuary is a significant data gap.

A third major data gap was that we only had adequate information to address four factors. The selection of these four factors was based upon the relative effects of each limiting factor within these two environments and the availability of empirical information to include in the evaluation. We did not select these four factors because we felt that they would have the most significant impact. Some of the factors not included may have a considerable affect on population viability of some ESUs. For example, we expect that estuarine water temperatures have warmed from historic levels which could affect metabolic processes of both salmon and their predators. These changes could increase mortality rates of salmon while in the estuary. Further, warmer water temperatures may exclude some habitat from use by juveniles during part of the year.

Northern Pikeminnow Predation on Juvenile Salmonids

As we noted previously, many predator prey interactions in the Columbia River estuary involving juvenile salmon have potentially changed from historic conditions. We treated predation by Caspian terns on juvenile salmonids as a “full” limiting factor because it met our criteria for inclusion. While other predators were not included because they did not meet our criterion, one predator that nearly had enough information to include as a limiting factor: northern pikeminnow. We include a discussion here of predation by northern pikeminnow as another example of the effects of a predator.

Northern pikeminnow are relatively large, long-lived, slow-growing, predaceous minnows native to the Columbia River basin and other parts of the Pacific Northwest. Maximum fork length, weight, and age are approximately 600 mm, 2.5 kg, and at least 16 years in the Columbia River (Parker et al. 1995). Large, old individuals typically dominate unexploited populations of northern pikeminnow. Juvenile salmonids are generally an important diet component only for these large, old northern pikeminnow (Vigg et al. 1991; Zimmerman 1999), and consumption rates of juvenile salmonids by northern pikeminnow increase as size increases. Zimmerman (1999) found that fish consumed by northern pikeminnow in all reaches of the Columbia River were overwhelmingly juvenile salmonids, and that numerical frequency of Chinook salmon greatly exceeded that of steelhead. Daily consumption of juvenile salmonids was greater in summer than in spring.

Development of the hydropower system in the Columbia River basin has resulted in increased losses of juvenile salmonids to northern pikeminnow. At dams, migrating juvenile salmonids are concentrated in forebays and tailraces, causing increased predation and salmonid loss (Poe et al. 1991; Vigg et al. 1991; Ward et al. 1995). Migration past dams also causes injury and physiological stress, which may increase the vulnerability of salmonids to predators (Mesa 1994). Impoundments increase travel time for migrating juvenile salmonids, prolonging their exposure to predators (Raymond 1988; Poe et al. 1991).

Significant losses of juvenile salmonids occur downstream of Bonneville Dam.. Beamesderfer et al. (1996) estimated that about 10 million juvenile salmonids were consumed annually by northern pikeminnow in the Columbia River downstream from Bonneville Dam prior to implementation of a Northern Pikeminnow Management Program (NPMP). This was about 5% of the approximately 200 million juvenile salmonid migrants in the Columbia River Basin, but a much higher percentage of salmonids that reached the river downstream from Bonneville Dam. This estimated loss exceeded the total estimate for the remainder of the lower Columbia and Snake rivers. Estimates of predation losses were relatively unbiased by consumption of juvenile salmonids killed by dam passage (Gadomski and Hall-Griswold 1992; Petersen et al. 1994). Abundance of northern pikeminnow downstream from Bonneville Dam greater than 250 mm fork length likely ranged from 600,000 to 800,000 individuals (Beamesderfer et al.1996; ODFW, unpublished data).

Unlike other resident fish predators, population dynamics and behavior of northern pikeminnow indicate that reductions in predation through a removal program are feasible. The NPMP began in 1990, based on findings from the earlier work conducted in John Day Reservoir (Rieman and Beamesderfer 1990; Poe et al. 1991; Vigg et al. 1991; Beamesderfer and Rieman 1991; Rieman et al. 1991). Since 1990, over 750,000 northern pikeminnow have been removed from the Columbia River downstream from Bonneville Dam. Abundance of large northern pikeminnow downstream from Bonneville Dam has been reduced to about 500,000 individuals (ODFW, unpublished data).Based on changes in the population structure of northern pikeminnow resulting from harvest, Friesen and Ward (1999) estimated that predation on juvenile salmonids by northern pikeminnow was reduced by approximately 25% annually.

Empirically derived estimates of predation indicate that reductions since implementation of the NPMP may be more substantial. Zimmerman and Ward (1999) estimated that predation from 1994-96 was about 50% of predation in 1992. Zimmerman et al. (2000) reported similar results for 1999.

The major uncertainties that precluded our including northern pikeminnow predation as a limiting factor were:

1. Uncertainty about historic levels of predation in this reach of the river. While the high levels of predation in the tailrace of Bonneville Dam is a change from historic levels, we could not determine whether predation downstream of this point had increased from historic levels.

2. Uncertainty about where predation occurs. While predation is generally regarded as most significant in littoral areas this is defined as <13m. We could not determine more specifically where predation in the littoral was occurring.

3. Uncertainty about what size classes were affected in the portion of the river. While Zimmerman (1999) provides data on size of salmonids eaten by northern pikeminnow throughout the basin, we could not determine what sizes of salmonids were being eaten in the river below Bonneville.

Direct Affects of the Hydropower System on the Estuary

One of the outcomes of this analysis is that we identified a clear linkage between hydropower operations and two of the factors we considered: flow and habitat. We did not consider the operation of the hydropower system to have a direct affect on either toxics or tern predation other than perhaps by increasing travel time and hence exposure of the fish to both predators and toxics. We concluded that most of the flow changes and some of the changes in habitat in the estuary could be directly attributed to affects (e.g., reduction in magnitude of the spring freshets) of the Federal Hydropower System.

For example, reductions in flow can permanently eliminate some habitat from use by estuarine dependent life history strategies. Even though the habitat may not be diked, it becomes functionally "too high" in elevation for the fish to use because of reductions in flow. In addition, because there is a relationship between flow and habitat, the value of some habitat can be diminished by a reduction in flow because it becomes accessible for less time than under historic conditions. Many attributes of the plume environment are also directly affected by the hydropower system including the size, shape, and seasonal movements.

While some of the affects of the hydropower system in the estuary could ***potentially*** be mitigated by altering flow regimes (e.g., increasing flows in spring and summer), there are clearly a variety of other consequences that would need to be considered with such changes (e.g., an increase in gas bubble disease). Some habitat changes not directly linked to affects of the hydropower system are possible, however. This involves restoring some of the shallow water habitat used by ocean type ESUs and the fry and fingerling portions of stream type ESUs that has been isolated from the system by levees and dikes. Dikes permanently isolate this habitat and make either direct use by the juvenile salmonids (access) or indirect use (organic matter transport) impossible. Breaching or leveling dikes is clearly a strategy that can be used to restore some of the shallow water habitat important to these shallow water dependent strategies.

How Much Change is Necessary to Affect Population Status

From the perspective of the estuary, population viability of stream type ESUs is most affected by tern predation and flow while ocean type ESUs are most affected by flow and habitat. At this time, we do not know how much of a change in each factor is required to affect viability of relevant ESUs. Probably the greatest opportunity to affect ESUs by manipulating one of these factors is by restoring lost, shallow water, low velocity, vegetated habitat (e.g., emergent marsh). This is because there is a strong linkage between dominant life history strategies of ocean type ESUs and shallow water habitat. A large amount of that habitat type has been lost due to diking. Clearly, restoration of some shallow water habitat can be done without changing hydro operations.

While more shallow water habitat could be made available with flow changes, this would have to be very carefully considered because it could have other, unintended consequences such as increase in gas bubble disease. We must also recognize that any questions about how much restoration is needed must address the question of how much change is possible. There are now constraints on the system such as climate change and "permanent" changes in the landscape such as from urbanization that clearly constrain how much change is possible; a return to pre-settlement conditions seems unreasonable to expect. We expect that studies now underway will provide more insight into how much change in shallow water habitat is both possible and needed.

Our analyses did not attempt to compare the ability of estuary factors and non estuary factors to affect viability. The analysis by Kaeriva et al. (2000) and more recent analyses by (McClure et al. in prep) suggest that the estuary and plume environment are generally important to the productivity of anadromous populations. In general, recovery is a life cycle process that requires strategies that focuses on crafting and evaluating alternative scenarios involving all life stages of the animals.

Thus, addressing estuarine factors can potentially improve population status and help recovery and should be included in any comprehensive plans for recovering populations in the Basin. A variety of factors (such as landscape connectivity within and between habitat zones) will need to be considered in deciding how to distribute recovery efforts directed at any suite of limiting factor. Any recovery actions directed at shallow water habitats will also need to consider the cumulative effects of all factors affecting ocean type populations in these habitats. For example, some recovery actions directed at habitat restoration in shallow water areas may need to simultaneously reduce toxic contamination if sites targeted for restoration are found to be contaminated.

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